UNIVERSITÉ DE GRENOBLE

THÈSE

Pour obtenir le grade de

DOCTEUR DE L'UNIVERSITÉ DE GRENOBLE

Spécialité : **Biodiversité-Ecologie-Environnement**

Arrêté ministériel : 7 août 2006

Présentée par

Pénélope LAMARQUE

Thèse dirigée par **Sandra LAVOREL**

préparée au sein du **Laboratoire d'Ecologie Alpine**
dans **l'École Doctorale Chimie et Sciences du vivant**

Une approche socio-écologique des services écosystémiques

Cas d'étude des prairies subalpines du Lautaret

Thèse soutenue publiquement le **11 juillet 2012**,
devant le jury composé de :

Michel DURU
Directeur de recherche à l'INRA, Toulouse, France (Rapporteur)
Pascal MARTY
Professeur à l'université de La Rochelle, France (Rapporteur)
Bas PEDROLI
Associate Professor à l'Université de Wageningen, Pays-Bas (Examinateur)
Ariane WALZ
Junior Professor à l'Université de Postdam, Allemagne (Examinatrice)
Sandra LAVOREL
Directrice de recherche au CNRS, Grenoble, France (Directrice de thèse)

Université Joseph Fourier / Université Pierre Mendès France /
Université Stendhal / Université de Savoie / Grenoble INP

Remerciements

On m'a souvent demandé : Une thèse … mais qu'est ce que c'est ? Après trois ans de réflexions dans le domaine, voici un bref aperçu de ma vision des choses.

Pour moi une thèse c'est comme partir à l'aventure pour une grande randonnée d'un peu plus de trois ans pour laquelle on a une petite idée de la région de destination et une vague idée de l'itinéraire. La fougue de la jeunesse m'a fait partir la fleur au fusil, testant de multiples chemins et batifolant dans les grands paysages. Rapidement, il y a eu des sommets à franchir, desquels d'autres se sont dessinés au loin. Mais des chemins plus plats, voire des descentes au fond des vallées ont régulièrement changé le rythme de la randonnée. Il y a des carrefours où j'ai été amenée à faire des choix parfois difficiles vu les intérêts présentés par les différents chemins. De temps à autres la présence d'un brouillard épais m'a fait perdre le sens de l'orientation, mais cela fait partie de l'apprentissage. S'il m'est arrivé de vouloir passer un peu plus de temps dans un coin agréable ou peu fréquenté, des événements climatiques imprévus de type tempête m'ont vite rappelé à l'ordre. Il m'est aussi arrivé de prendre des chemins glissants ou des voies sans issue mais j'ai toujours fini par retomber sur mes pattes et finalement un beau jour la destination finale s'est dessinée à l'horizon et j'ai alors entamé un sprint final, heureuse d'arriver au bout de cette belle aventure. Si vous souhaitez parcourir plus en profondeur certaines étapes de cette randonnée, vous trouverez quelques topo-guides aux éditions Springer et Elsevier que j'ai pu rédigés à l'ombre d'un grand pin ou au coin d'une cheminée. Que cela reste entre nous, mais finalement après ces trois années parcourues, ce qui reste en mémoire, ce sont évidement tous ces beaux paysages, ces moments de découverte, mais surtout les innombrables rencontres réalisées tout au long du chemin. Sans elles, il me serait probablement arrivé de prendre un mauvais chemin, de trouver le temps long ou de perdre courage pour arriver à destination. En effet, une thèse n'est pas un travail purement personnel, elle se construit autour des discussions, des idées et des conseils des uns et des autres. C'est pourquoi, je voudrais dire à toutes ces personnes rencontrées, ainsi qu'aux personnes qui m'ont soutenue dans ce projet. Un énooooorrrmmme ***MERCI à vous tous*** *!*

En fait, c'est en 2006 qu'a commencé l'aventure de ma thèse. Encore étudiante en master en géographie à l'UCL en Belgique, j'étais loin de me douter que le regard porté sur une couverture de thèse déposée sur le bureau de mon encadrant (Eric Lambin) allait m'emmener si loin. Cette thèse était celle de **Fabien Quétier** (Si on m'avait dit qu'un jour j'allais travailler avec toi … tu n'y étais pas pour grand-chose mais déjà merci pour ca !). Curieuse, j'ai demandé à l'emprunter pour la parcourir et c'est de cette manière que j'ai fait connaissance avec les recherches menées au LECA et les services écosystémiques ! Internet, m'a ensuite permis de faire des recherches plus poussée sur les travaux d'une certaine Sandra Lavorel auxquels j'ai porté beaucoup d'intérêt. J'ai mis tout cela dans un coin de ma tête car je voulais absolument tenter ma propre expérience dans le monde professionnel autre que le monde académique et de la recherche. Mais après deux années, bien des aspects de la recherche me manquaient et tout cela est revenu trotter dans ma tête. Timide de nature (si si croyez moi), je n'osais pas faire le premier pas et contacter Dr. Sandra Lavorel dont plusieurs personnes m'avaient fait le portrait d'une grande chercheuse en me ventant la qualité de ses recherches et son implication dans de nombreux projets (ils n'avaient pas tort). C'est finalement début 2008, sous les conseils d'**Eric** (merci !), qui m'a décrit Sandra comme une personne accessible ayant de nombreuses qualités humaines (ce qui s'est révélé vrai après avoir appris à la connaitre), que le cœur battant à 100 à l'heure et les mains moites j'envoyais un mail à Sandra pour lui demander si elle avait des sujets de thèse à proposer. J'ai rapidement eu la chance d'avoir un entretien avec elle, qui m'a conforté dans mon envie de travailler au LECA dans son équipe. La motivation étant apparemment réciproque, un peu plus de 6 mois après j'étais engagée au LECA en tant qu'ingénieur d'étude pour faire la cartographie des services écosystémiques avant de poursuivre en thèse 5 mois plus tard au sein du projet VITAL qui avait entre temps été accepté. C'est là que l'aventure a commencé pour de bon, m'emmenant tout de suite *dré dans l'pentu*.

Sandra, je voudrais donc te remercier de m'avoir permis de me lancer dans cette aventure. Malgré tes nombreuses responsabilités et ton planning chargé tu t'es toujours rendue disponible pour m'aider et m'encadrer au mieux tout en laissant exprimer ma créativité. Dès le début tu as cru en moi et tu n'as jamais douté dans mes capacités à mener à bien mon travail alors que moi-même je n'y croyais parfois plus, et encouragé à poursuivre mes travaux toujours plus loin en rehaussant toujours plus la barre de tes exigences. Tu as mis à ma disposition les moyens nécessaires (parfois considérables) pour mettre en œuvre mes idées, aller les partager avec d'autres personnes pour les affiner ou les poursuivre et me redonner confiance dans l'intérêt de mes recherches. Je remercierai donc en passant toutes les personnes rencontrées durant les

colloques, les écoles d'été, les workshops qui par leurs expériences m'ont permis d'enrichir la mienne. Sandra, bien qu'il me soit arrivé de te critiquer pour l'une ou l'autre petite chose (un chef est toujours critiqué à un moment ou un autre), de prendre la mouche à un e-mail mal interprété, travailler avec toi a été très enrichissant et très agréable. J'espère que nous aurons d'autres occasions de collaborer ! Trouver les bon mots étant difficile, ce mot magique résume sans doute beaucoup de chose : MERCI pour tout !

Mon fort attachement aux Bauges étant quelque chose auquel il est difficile de me faire renoncer, comme beaucoup d'entre vous s'en seront aperçus, Sandra m'a permis de rester dans le secteur (merci !) en me laissant intégrer la petite **équipe de Chambéry** qui m'a accueillie comme dans une petite famille. Merci à **Christiane, Claude, Antoine, Tony, Claire, Anne, Annie, François, Dave, et tous les autres** pour votre accueil, votre enthousiasme, les bons gâteaux et les barbecues aux Mottets ! Ce n'est pas sans regrets qu'un an plus tard je laissais mon bureau de Chambéry pour essayer de me reprocher de l'équipe de Grenoble. Sandra m'a rapidement fait une petite place dans son bureau. Merci **Sandra** d'avoir été une sympathique colloque de bureau ! C'est alors qu'a commencé une série d'aller-retour entre les Bauges et Grenoble, heureusement **Christian et Jacky** étaient là pour animer les longs trajets de train et m'empêcher de continuer à travailler pendant ce temps. Merci à vous pour votre bonne humeur !

J'ai petit à petit appris à connaitre les membres de l'équipe TDE devenue FYSE et ensuite les autres membres du labo (c'est qu'à Grenoble il y a beaucoup plus de monde qu'à Chambéry alors au début ca fait presque peur ;-)). Pour être sûre de n'oublier personne, je vous dis d'ores et déjà **merci à tous** pour votre accueil, les bavardages dans les couloirs ou lors des *tea-time* (merci aussi aux organisateurs du **bureau 111**) et les sorties ski.

Merci aussi à mes **collègues de la FYSE/TDE Team** avec qui j'ai eu l'occasion de travailler ou de discuter de choses scientifiques… ou non, de se serrer les coudes pendant les moments difficiles (**MP, JC, Jacky, Nico, Jonath, Karl, Cindy, Marie-Lise, Patrick**…). Un merci tout particulier à **Maud et Pierre** qui ont eu la patiente de m'expliquer R dans la dernière ligne droite de la thèse. Ce n'était pas une tâche facile ! Merci aussi à **Maud** pour son soutien moral les dernières semaines de rédactions et à **MP** qui veille toujours d'un œil sur nous telle une petite maman. Merci **JC** d'avoir eu la patiente d'écouter ou en tout cas faire *semblant d'écouter mes longs discours pendant mes brèves interruptions dans ton bureau.*

*Merci à ceux d'entre vous qui m'ont accompagné au cours de différents voyages … heeuuu pardon … de déplacements professionnels ! (**Tony, Fabien, Cécile, Isabelle, Nico** (merci de m'avoir fait découvrir les nuits autrichiennes différemment ;-)), **JC, Bello, Karl, Sandra**).*

*Lorsque je suis arrivée au LECA, mon sujet de thèse intriguait bien des personnes. En effet, quelle idée de faire pour terrain des enquêtes auprès d'éleveurs ou de gestionnaires au lieu de passer des heures à compter les fleurs ou trier la biomasse ! Heureusement, **Fabien Quétier** (un type génial, si si on peut le dire) était déjà passé par là et a tout de suite consacré du temps à répondre par de long mail à mes interminables questions alors qu'il était encore de l'autre coté de la terre. Son retour au LECA à bien facilité les choses puisque les mails ont été remplacés par de nombreuses discussions et remises en questions de mes hypothèses qui m'ont amené à m'interroger plus profondément sur le concept de services écosystémiques. Merci Fabien pour ton sens de l'animation de discussion, du démontage d'arguments longuement réfléchis et surtout pour ton aide précieuse et ton soutien tout au long de la thèse. Petit à petit, l'arrivage de nouveau thésards et post-doc a amené du sang neuf dans nos discussions et a permis la création d'un petit groupe restreint de personnes travaillant sur ce concept farfelu des services des écosystèmes (par ordre d'apparition : **Pierre, Maud, Coline, Emilie**). De temps en temps, **Nico** a aussi eu le courage de ce greffer à ce petit groupe, le temps d'un journal club à rallonge. Merci à tous pour les intéressantes discussions qu'on a pu avoir sur le sujet.*

*L'avantage de faire une thèse dans un projet international c'est de rapidement travailler avec de multiples partenaires. Thank you to all **VITAL partners** and especially **Ulrike, Richard, Melanie, Markus, Catherine and Ute**, for your help to write my first paper, your advices and encouragements.*

*A son image Sandra m'a rapidement appris à jongler entre différent projets et rencontrer d'autres collaborateurs. J'ai donc également eu la chance de prendre part au projet **Secalp** et tout particulièrement de faire la connaissance et de travailler avec **Baptiste Nettier et Laurent Dobremez**. Merci à vous pour vos précieux conseils et une très sympathique collaboration.*

Les collaborations ne se sont pas arrêtée aux frontières des ces deux projets. Merci aussi au **Parc National des Ecrins (Richard, Muriel, Cédric), au CERPAM (Simon) et la Chambre d'Agriculture (Jean-Luc)**. Merci aussi à **Cécile Barnaud, Adrienne Gret-regamey, Philippe Fleury, Thomas Koellner, Eric Lambin, Patrick Meyfroidt et Fabien Quétier** d'avoir accepté de faire partie de mon **comité de thèse**. Par vos nombreux conseils vous m'avez permis d'avancer chaque année un peu plus près du but. Merci !

A peine la thèse commencée, j'accompagnais Eric Deboeuf, **stagiaire** du projet SECALP dans mes premières enquêtes avec les éleveurs de Villar d'Arène. D'autres stagiaires m'ont ensuite accompagnée pour certaines enquêtes, ateliers ou jeux avec les éleveurs ou d'autres acteurs ou pour m'aider à réaliser les tâches les plus ingrates de ma thèse. Merci aux stagiaires (**Eric, Andrea, Maud, Pierre, Aloïs** (je n'ai pas oublié le tréma) et **Claire**) qui ont fait un bout de chemin avec moi, pour votre aide précieuse, les bons moments passés ensemble et pour votre indulgence face à mes premiers pas d'encadrante.

Si j'ai rencontré pour la première fois les **éleveurs** au printemps 2009, tout comme moi, ils étaient loin de se douter qu'on allait se rencontrer si souvent au cours des trois années suivantes. Mais comment ne pas avoir envie de poursuivre un travail avec des éleveurs aussi passionnés par leur travail et aussi captivants dans leur façon d'en parler. Merci à vous, les éleveurs faranchins (**André, Bertrand, Jean-françois, Marie-Cécile, Michel, Maurice, Raymond, Fredéric**) ou de l'autre coté du tunnel (**Eric**). Sans vous cette thèse n'aurait pas pu être réalisée. Avec toute ma sympathie, je vous dis merci pour tous ce que vous m'avez appris sur l'agriculture de montagne, pour le temps précieux que vous m'avez consacré et de vous être prêtés au jeu de mon étude et des recherches du LECA. Merci aussi à l'un d'entre vous (qui j'espère ce reconnaitra) qui m'a un jour dit quelque chose qui a dû me trotter dans un coin de la tête : « il faut aller là où les autres ne vont pas ! ».

Merci à **toutes les autres personnes interviewées**, dont la liste serait trop longue pour être énumérée ici, pour le temps consacré et la richesse des discussions.

Si la bonne réalisation d'enquêtes repose généralement sur la mise en place d'un climat de confiance et d'une bonne connaissance des acteurs, c'est à la **SAJF** et plus particulièrement à **Serge** qu'il faut que je dise merci pour le formidable travail qu'il réalise avec les habitants de Villar d'Arène. Merci car sans l'intermédiaire du Jardin Alpin, il serait probablement bien plus difficile de communiquer nos travaux de recherches avec les habitants et partenaires locaux. Une thèse passe si vite que je n'ai malheureusement pas eu le temps de bien connaitre tout les membres de la SAJF malgré la proximité de vos bureaux et du mien … Roland, Pascal, Eliane, Joëlle, Stéphane …. dommage qu'il n'y ait personne en automne ou au printemps au Lautaret … C'est peut-être ça qu'on veut dire quand on dit qu'il y a compromis entre agronomie et écologie c'est qu'en fait on n'utilise pas les même lieux durant la même fenêtre temporelle ? Heureusement qu'il y a le tournoi de pétanque de la SAJF, merci **Roland** pour l'organisation de ce sympathique tournoi !

Au cours d'une thèse, il y a aussi pas mal d'aspects logistique à gérer. Ceci a pu être réalisé grâce aux filles du troisième étage **Kim, Florence, Nancy, Joelle**. Merci pour la réalisation des nombreux ordres de missions et bons de commande, et votre bonne humeur quotidienne.

S'il y a bien quelque chose qui m'a accompagné quotidiennement pendant toute la thèse c'est mon **ordinateur** ! Je ne sais pas si c'est à force de surmenage ou d'énervement de m'a part, mais en « bugant » il m'a à plusieurs reprise obligé à prendre une petite pause. Heureusement, Olivier à toujours été là et grâce à sa patiente à réussi à le refaire fonctionner. Merci **Olivier** !

Finalement, ces riches rencontres se sont terminées par le face à face avec les **membres de mon Jury**, plutôt éclectique vu la nature interdisciplinaire de ma thèse. Merci à **Michel Duru** et **Pascal Marty** pour le temps consacré à la rédaction de leurs rapports, ainsi qu'a **Ariane Walz** et **Bas Pedroli** qui ont consacré du temps à lire et évaluer ma thèse à cheval sur des disciplines qui n'était pas toujours proche de la leur. Merci pour vos commentaires et vos encouragements qui m'ont mis en confiance pour la suite.

Et comme ces pages constituent l'une très trop rare occasion où l'on prend le temps de dire merci. J'en profiterai pour balayer un rayon de connaissances un peu plus large que celles réalisées durant ces trois années de thèse. Finalement, une thèse on ne décide pas de la faire sur un coup de tête, c'est quelque chose qui muri lentement dans l'esprit … Merci donc à **Isar, Isabelle et Maud** qui m'ont permis de passer la difficile étape de la première candi. Merci aussi aux **géographes** louvanistes qui m'ont donné envie de poursuivre dans cette belle discipline.

Merci à mes amis agriculteurs qui m'ont ouvert les yeux vers une autre montagne que celle du sport et m'ont donné l'envie de me passionner pour cette utilisation traditionnelle à l'interface entre homme et nature. Dans le prolongement merci aux personnes du SUACI et du PNR du Massif des Bauges qui m'ont permis de faire mes premiers pas professionnels dans le monde agricole et de poursuivre mon apprentissage dans le domaine.

Merci à **mes amis baujus**. Céline pour ton amitié, ta patience, ton écoute (merci aussi d'avoir mis tes talents d'artiste à contribution de la science pour la réalisation des pions du jeu en un temps record !!!) , Roland de m'avoir appris tant de chose sur la vie d'un vrai baujus et de m'avoir déconnecté de mon boulot aux moments où j'en avais besoin, Michel (« quand on veut on peut ! » tu avais raison), Michel et Michel, Philippe, Philippe et Philippe (non j'ai pas abusé d'alcool baujus… mais il y a des prénoms plus courant que d'autres), Françoise, Valérie, Émilie, Arthur, Jean-Luc, Sophie, Sophie, Lilian, Myriam, Ghis, Benoit, Radio Alto, ….).

Alexis, il y aurait tant de choses pour lesquelles te remercier que je ne sais pas par où commencer … alors en attendant merci de tout cœur d'avoir été auprès de moi tout au long de ces années.

Mes parents. Pas vraiment pour la thèse mais pour avoir toujours cru en moi et m'avoir permis de toujours faire ce que j'avais envie.

Ma petite étoile qui me guide au quotidien.

A cette vieille grange au milieu des prés qui m'a donné l'inspiration pendant la seconde moitié de la thèse.

Table des matières / Contents

Avant propos

Cette étude s'inscrit dans le cadre de deux projets de recherche menés par le Laboratoire d'Ecologie Alpine, Grenoble portant chacun sur les services des écosystèmes et leurs évolutions dans un contexte de changements globaux. Le premier aborde le sujet sous l'angle de l'écologie fonctionnelle, le second par une approche agronomique.

Le projet VITAL: "Ecosystem service provision from coupled plant and microbial functional diversity in managed grasslands"

Ce projet international et multidisciplinaire est financé dans le cadre de l'EraNet-BiodivERsA (http://www.eurobiodiversa.org). Il a pour objectif de produire un modèle conceptuel des relations entre la biodiversité du sol, des plantes et les services des écosystèmes par une approche traits fonctionnels dans les prairies de montagnes. Ces relations sont étudiées dans le contexte du changement global par l'intermédiaire de scénarios prenant en compte l'effet direct de la gestion des prairies, et l'effet indirect du climat, des politiques publiques, des marchés et de la gouvernance. A cette fin, VITAL propose de répondre à six objectifs faisant chacun l'objet d'un axe de recherche : (1) identifier les principaux services des écosystèmes ; (2) identifier les indicateurs des mécanismes sous-tendant les réponses des plantes et du sol aux changements de gestion, (3) développer un modèle conceptuel des réponses en terme de diversité fonctionnelle (sol et plantes) face aux changements de pratiques, (4) valider le modèle sur des zones d'étude présentant un gradient d'intensité de pratiques, (5) projeter la fourniture de services des écosystèmes dans le futur selon différents scénarios, (6) identifier et répondre aux besoins des acteurs locaux concernés et des décideurs politiques. Cette approche est testée sur trois sites d'étude fournissant une gamme représentative des tendances de gestion agricole typiques des régions montagneuses de l'Europe de l'Ouest : Alpes françaises (Lautaret), autrichiennes (Stubai Valley), et montagnes du nord de l'Angleterre (Yorkshire Dales). Mon étude a porté particulièrement sur les objectifs 1, 5 et 6 menés sur le site français. Ce projet, porté par différentes universités, m'a conduite dans le cadre de cette thèse à travailler en étroite relation avec Ulrike Tappeiner (écologiste du paysage), Markus Schermer et Melanie Steinbacher, (sociologues) à l'Université d'Innsbruck (UIBK, Austria), Richard Bardgett et Catherine Turner (écologistes du sol) de l'Université de Lancaster (UK).

Le projet SECALP: « Adaptation des territoires alpins à la recrudescence des sécheresses dans un contexte de changement global »

Le projet SECALP est un projet français financé dans le cadre de l'appel à projet GICC-MEDDTL impliquant des chercheurs du Laboratoire d'Ecologie Alpine et de l'Irstea. Son objectif est d'analyser les mécanismes d'adaptation des territoires semi-naturels de montagne (le Vercors et le Lautaret) face aux changements climatiques, particulièrement la récurrence des sécheresses.

SECALP est articulé autour de quatre axes de recherche complémentaires et interactifs, répondant chacun à des objectifs spécifiques: (1) identifier les mécanismes écologiques d'adaptation aux changements climatiques, dont la sècheresse, (2) comprendre les processus d'adaptation des acteurs utilisateurs de l'espace, (3) proposer des orientations et des leviers pour une gestion durable des espaces naturels de montagne, (4) proposer des stratégies d'observation à long terme intégrant le changement climatique. Cette thèse s'inscrit principalement dans les axes 2 et 3, en collaboration avec deux agronomes de l'Irstea Grenoble : Baptiste Nettier et Laurent Dobremez.

Chapitre 1

Introduction

Ce premier chapitre présente la structure générale de la thèse. La première partie expose le contexte et introduit les recherches actuelles dans le domaine des services écosystémiques. La deuxième partie expose les objectifs et les hypothèses de la thèse. Finalement, ce chapitre conclut par une description de la zone d'étude dans son contexte général et exprime les motivations qui ont conduit à choisir cette zone comme cas d'étude.

1 L'émergence d'un concept : les « services écosystémiques »

L'entrée de notre civilisation dans une nouvelle Ere géologique appelée « Anthropocène » (Steffen et al., 2007) décrit le passage de l'adaptation de l'Homme à l'environnement à l'adaptation de l'environnement lui-même à la présence de l'Homme. En effet, la croissance exponentielle des activités humaines est devenue le principal facteur de changement du système terrestre (Steffen et al., 2007). L'augmentation de cette pression humaine sur la planète pourrait déstabiliser de manière critique les systèmes biophysiques et causer des changements environnementaux irréversibles (Rockstrom et al., 2009). Rockstrom et al. (2009) démontrent que trois processus actuellement à l'œuvre - changement climatique, perte de biodiversité accélérée et interférence avec le cycle de l'azote – ont déjà dépassé les limites définissant un espace de manœuvre au sein duquel l'homme peut vivre en adéquation avec les systèmes et processus biophysiques. L'accélération de la perte de biodiversité a franchi cette limite principalement à cause des changements d'utilisation du sol (Sala et al., 2000). A ce jour, le taux de perte de biodiversité est estimé 100 à 1000 fois supérieur à ce qui pourrait être considéré comme naturel (Rockstrom et al., 2009). En outre, il a aussi été clairement démontré que la perte de biodiversité peut affecter les propriétés des écosystèmes et le bien-être humain (Diaz et al., 2006; MEA, 2005).

En l'absence de changements majeurs des comportements humains et des politiques publiques, les effets des activités humaines sur l'environnement vont continuer d'altérer la biodiversité (Chapin et al., 2000). La conservation de la biodiversité pour sa simple valeur morale, la préservant pour elle-même, s'est avérée insuffisante pour arrêter ou même diminuer la perte de biodiversité face aux impératifs sociaux et économiques croissants (Mace et al., 2010). Face à ce constat, bien que l'identification et la reconnaissance des biens et services que la population humaine reçoit de la nature ne soient pas nouvelles, une attention croissante est portée depuis à la biodiversité dans cette acceptation, sous l'appellation services des écosystèmes ou services écosystémiques (« ecosystem services » en anglais) (MEA, 2005) (Voir encadré 1). Ce concept de services des écosystèmes fournit une nouvelle justification anthropocentrique de conservation des espèces et des écosystèmes, basée sur notre dépendance à l'égard des biens et services qu'ils nous fournissent. En plus d'être largement utilisé pour attirer l'attention sur la biodiversité, il a très vite servi de support théorique à des études sur les relations entre la biodiversité à différentes échelles et les sociétés qui en dépendent (Martinez-Harms and Balvanera, 2012; Vihervaara et al., 2010).

Le concept de services des écosystèmes est apparu en premier chez les biologistes de la conservation tels que Ehrlich and Mooney (1983) afin d'attirer l'attention au niveau mondial sur la perte de biodiversité et la dégradation des écosystèmes. Dans la foulée de la publication du livre de Daily (1997) sur les services de la nature, plusieurs conférences internationales et conventions ont accordé une importance particulière aux services des écosystèmes dans leur programme (ex. Convention on Biological Diversity (CBD), Ramsar, IUCN World Conservation Congress). En 2005, la diffusion du rapport d'évaluation des écosystèmes pour le millénaire (Millennium Ecosystem Assessment (MEA, 2005)) a constitué une étape décisive dans la promotion du concept (Seppelt et al., 2011; Vihervaara et al., 2010), soulignant la dépendance du bien-être humain envers les écosystèmes en insistant sur la nécessité de mieux décrire, quantifier et évaluer écologiquement, culturellement et économiquement leur importance. Ceci à conduit le programme international DIVERSITAS sur les sciences de la biodiversité à placer les services des écosystèmes au cœur d'un de ses quatre projets principaux (ecoSERVICES) et à appuyer la création de la plateforme intergouvernementale sur la biodiversité et les services des écosystèmes (IPBES) qui, à l'image du GIEC (Groupe d'experts intergouvernemental sur l'évolution du climat), vise à rapprocher les scientifiques des décideurs politiques en donnant une réponse à la fois locale et globale à l'érosion de la biodiversité (Perrings et al., 2011). Le succès du concept dépasse largement la biologie de la conservation se propageant dans de nombreuses autres disciplines y compris les sciences sociales et économiques et ainsi que dans des sphères non scientifiques (ex. UK National Ecosystem Assessment, 2011). L'évaluation économique des écosystèmes existait déjà bien avant l'émergence du concept (ex. (Krutilla, 1967; Westman, 1977)) mais son importance a considérablement augmenté depuis la parution de l'article de Constanza et al. (1997) attribuant une valeur monétaire (exprimée en dollars) aux services des écosystèmes mondiaux (voir Gómez-Baggethun et al., 2010 pour une revue). En 2008, une étude mondiale sur l'économie des écosystèmes et de la biodiversité (TEEB) a été lancée par le G8 et cinq grands pays émergents pour promouvoir l'intégration des valeurs économiques de la biodiversité et des services rendus par les écosystèmes dans les processus de prise de décision et évaluer les conséquences de la perte de la biodiversité en terme de coûts pour l'économie mondiale (Ring et al., 2010).

Encadré 1 : Les classifications des services

L'évaluation des écosystèmes pour le millénaire (2005) a proposé une classification des services écosystémiques en quatre grandes catégories couramment reprises dans la littérature :

- Les **services d'approvisionnement** (*provisionning services*) sont les produits et matériaux obtenus des écosystèmes : eau, nourriture, fibre, ...
- Les **services de régulation** (*regulating services*) représentent les bénéfices tirés de la régulation des processus naturels : régulation du climat, régulations des cours d'eau, contrôle des maladies et ravageurs, ...
- Les **services culturels** (*cultural services*) font référence aux bénéfices non matériels apportés par les écosystèmes et les paysages : esthétique, lieux de loisirs
- Les **services de soutien** (*supporting services*) sont les services nécessaires à la production des autres services : formation des sols, photosynthèse, recyclage des nutriments et de l'eau

D'autres classifications plus opérationnelles ont été proposées telle que la classification proposée par Zhang (2007) distinguant trois catégories de services selon leurs rôles pour les agro-écosystèmes. Cette classification a été reprise et adaptée par l'expertise scientifique de l'INRA sur l'agriculture et la biodiversité (Le Roux et al., 2008). Elle distingue :

- Les **services intrants**, qui contribuent à la fourniture de ressources et au maintien des supports physico-chimiques de la production agricole, et qui assurent la régulation des interactions biotiques, positives ou négatives : maintien de la structure ou de la fertilité des sols, pollinisation, protection de la santé des animaux domestiques par exemple ;
- Les **services de production contribuant au revenu agricole :** ils concernent la production végétale et la production animale, dont les niveaux mais aussi la stabilité dans le temps et la qualité des produits sont considérés ;
- Les **services produits hors revenu agricole direct**, qui incluent le contrôle de la qualité des eaux, la séquestration du carbone ou la valeur esthétique des paysages notamment.

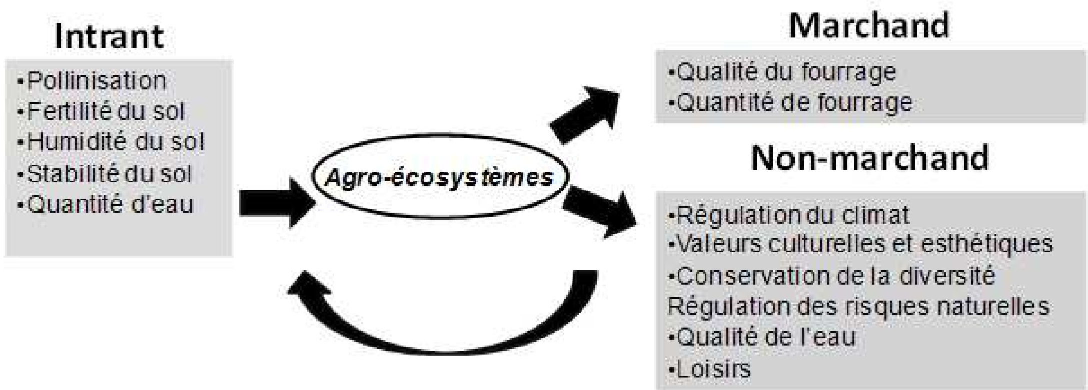


Schéma conceptuel de l'organisation des services pour et par les agro-écosystèmes. Exemple de services pour le cas de prairies. Adapté de Le Roux et al, 2008.

Cette intensification des recherches sur la valeur monétaire des écosystèmes à fait naitre un intérêt pour la conception d'instruments financiers axés sur l'incitation économique pour la conservation. Le concept de services des écosystèmes semble avoir aussi stimulé un changement d'orientation d'autres concepts. En agronomie par exemple, où le concept de « multifonctionnalité » de l'agriculture qui souligne le rôle non-marchand de l'agriculture (aménités et externalité positives) en considérant tous les biens, produits et services créés par les activités agricoles (Marsden and Sonnino, 2008), s'est orienté vers la multifonctionnalité des agro-écosystèmes (Simoncini, 2009), ouvrant cette perspective multifonctionnelle à la biodiversité. Le concept de service est d'ailleurs parfois repris dans le domaine de l'agriculture, de la forêt ou des ressources naturelles. Les services écosystémiques ont favorisé aussi l'émergence du concept d'intensification écologique de l'agriculture désignant le recours à la régulation biologique des agro-écosystèmes pour atteindre deux objectifs apparemment opposés : un niveau élevé de production alimentaire et fournir des services écosystémiques (Doré et al., 2011).

2 L'approche socio-écologique des services écosystémiques

Le concept de services des écosystèmes défini comme les bénéfices que l'homme retire des écosystèmes (MEA, 2005) joue un rôle charnière entre la biodiversité et le bien-être humain (Figure 1). Dans cette chaîne conceptuelle, les propriétés d'un écosystème regroupent ses structures (ex. les espèces, leurs abondances) et les processus écologiques (interactions entre espèces ou entre compartiments de l'écosystème tel que le sol et la végétation) qui sous-tendent sa capacité à offrir un ou plusieurs services. Cette capacité n'est que potentielle (appelée fonctions des écosystèmes) tant que ces services ne sont pas utilisés, consommés ou valorisés par des individus ou des groupes humains (Wallace, 2007). Les services écosystémiques se distinguent des attentes ou des besoins humains proprement dit (les bénéfices), puisque ceux-ci résultent généralement d'une combinaison de capitaux financiers, techniques et humains en sus de la contribution strictement liée aux processus écologiques (Boyd and Banzhaf, 2007).

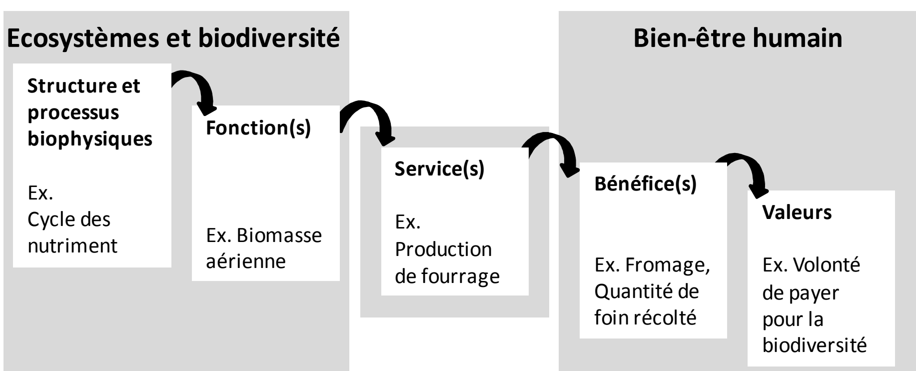


Figure 1 : Cascade conceptuelle illustrant le rôle charnière des services des écosystèmes dans chaîne d'interrelations entre l'écosystème et le bien-être humain (adapté de Haines-Young and Potschin, 2010).

Le concept de services étant à l'interface entre l'homme et son environnement, l'approche socio-écologique (Figure 2) a été proposée comme cadre conceptuel d'étude sur les services écosystémiques (Chapin et al., 2000; Collins et al., 2011; Diaz et al., 2011; Koellner, 2008; MEA, 2005; Stevenson, 2011). Ces systèmes complexes impliquant des éléments écologiques (biophysiques) et humains (sociétaux) et leurs interactions mutuelles sont maintenant décrits comme des systèmes couplés homme-environnement (coupled human and natural system (CHANS)) (Liu et al., 2007c) ou des systèmes socio-écologiques (social-ecological systems (SES)) (Ostrom, 2009). Le système écologique ou écosystème est défini comme « des complexes dynamiques composés de plantes, d'animaux, de micro-organismes, et de la nature morte environnante agissant en interaction en tant qu'unité fonctionnelle (MEA, 2005)». Le système social inclut l'économie, les hommes, les institutions et leurs interactions mutuelles (Harrington et al., 2010). Les socio-écosystèmes impliquent non seulement l'étude de la dynamique au sein d'un sous-système mais aussi l'étude de la complexité des interactions réciproques et de rétroactions entre écosystèmes et systèmes sociaux (Folke, 2006) au travers des structures et processus émergeant à différentes échelles (Liu et al., 2007a). Les systèmes sociaux et écologiques ne sont pas simplement liés, ils sont interconnectés ; leurs relations sont basées sur un partenariat mutuel et plutôt que sur une domination de l'un sur l'autre (Harrington et al., 2010). Ces études requièrent donc non seulement de considérer des variables écologiques (ex. biodiversité, habitat, cycle des nutriments) et des variables humaines (ex. processus socio-économiques, réseaux d'acteurs, gouvernance) mais aussi des variables qui lient les composantes écologiques et humaines (ex. utilisation des services écosystémiques, collecte de fourrages) (Liu et al., 2007b). Ce type d'étude intégrative mène à la réalisation de recherches interdisciplinaires impliquant différentes disciplines académiques et intégrant les connaissances de chacune d'elles dans un but commun et afin de créer de nouvelles connaissances et développer des théories propres (Tress et al., 2005). La prise en compte d'un éventail plus large de questions et l'utilisation d'approches incorporant simultanément de nombreux facteurs sont nécessaires afin de trouver des solutions durables aux problèmes environnementaux (Lambin et al., 2001).

Cependant, malgré une convergence de cadres conceptuels faisant référence à la relation entre les activités humaines et leurs effets sur les systèmes écologiques, la biodiversité et le bien-être humain (Stevenson, 2011; vanWey et al., 2005), les recherches sur les services se focalisent généralement sur l'un des systèmes et tendent à ignorer les rétroactions au sein de l'un ou l'autre des deux sous-systèmes et entre ces deux sous-systèmes, social et écologique (Nicholson et al., 2009). Les études en sciences naturelles se focalisent sur le sous-système écologique et la fourniture potentielle de services écosystémiques tandis que les études en sciences humaines se concentrent sur le sous-système social et la demande en services (Veldkamp, 2009). Néanmoins, il semblerait qu'une approche judicieuse soit de commencer par l'évaluation sociale avant de se pencher sur l'évaluation écologique des services, afin d'identifier les bénéficiaires des fonctions écologiques (Cowling et al., 2008).

Les concepts et approches que j'ai utilisés pour étudier les services écosystémiques sont décrits dans les sections ci-dessous pour chacune des composantes du système socio-écologique et leurs interrelations.

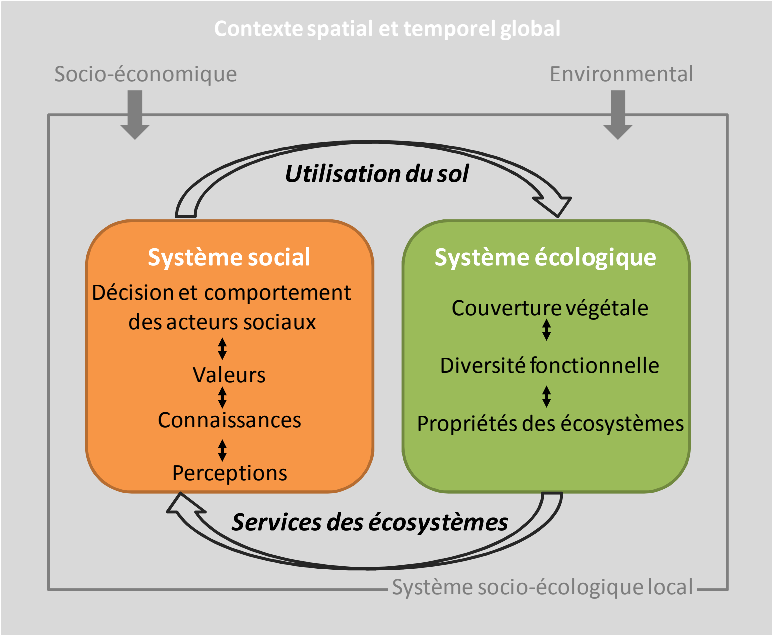


Figure 2 : Cadre conceptuel interdisciplinaire de la thèse liant la diversité fonctionnelle et le comportement des acteurs sociaux par l'utilisation du sol et les services des écosystèmes à une échelle locale. Adapté de Diaz et al.(2011) et Moran et al. (2005).

2.1 Le système social

Dans le domaine de recherche des services écosystémiques peu d'études ont porté sur la perception et sur les connaissances des services par les *stakeholders*[1] (Lewan and Soderqvist, 2002; O'Farrell et al., 2007; Pereira et al., 2005; Quétier et al., 2010b), notamment sur les connaissances des relations entre services. Pourtant, les *stakeholders* pouvant être des bénéficiaires de services, les recherches devraient être inspirées par ceux-ci afin de transmettre des informations utiles telles que l'identification des services dont ils bénéficient et considèrent importants pour leur bien-être (Cowling et al., 2008). C'est pourquoi une approche ascendante des *stakeholders* aux scientifiques est préconisée par rapport à une approche descendante. De

[1] Dans le domaine des services écosystémiques, les *stakeholders* sont des individus ou des groupes d'individus qui bénéficient ou qui affectent de manière passive ou active un ou plusieurs services. Adapté de Reed et al (2009). La traduction française de ce terme est problématique car elle dépend du domaine d'étude. Dans le domaine agricole et environnemental, il est souvent traduit par « porteur d'enjeux ».

plus, il semble utile d'inclure une participation précoce et explicite de ces *stakeholders* dans les projets de recherche afin d'en saisir la dimension humaine (Menzel and Teng, 2009). Toutefois, des études explorant les connaissances et perceptions par différents types de *stakeholders* autour de la biodiversité (Buijs et al., 2008; Fischer and Young, 2007), de l'utilisation des plantes (Pieroni and Giusti, 2009), de l'influence de la flore sur l'esthétique (Lindemann-Matthies et al., 2010) mais aussi sur le sol (Barrera-Bassols and Zinck, 2003), ont montré l'importance d'utiliser le savoir de la société. Un domaine de recherche sur les connaissances écologiques traditionnelles (Traditional ecological knowledge, TEK)) ou locales (Local ecological knowlege, LEK) se focalise sur la compréhension des relations qu'ont diverses sociétés humaines avec leur environnement et les connaissances inhérentes qui en découlent notamment dans une situation de changement des ressources naturelles (Berkes et al., 2000; Cheveau et al., 2008).

Actuellement, la demande en services écosystémiques est plutôt étudiée sous l'angle des valeurs qui leurs sont attribués par les *stakeholders*. Les valeurs sont des normes qui permettent de juger, individuellement ou collectivement si quelque chose (dans notre cas une fonction écosystémique) est par exemple bon, beau, vrai, utile ou moral. Ces valeurs peuvent être approchées de manière objective en essayant d'établir une hiérarchie entre les services ou de manière subjective en examinant la valeur d'un service par rapport à leur désirabilité relative (Salles, 2011). Les différentes valeurs que les *stakeholders* peuvent attribuer à la biodiversité et aux écosystèmes sont classés en deux grandes catégories : (i) les valeurs intrinsèques ou inhérentes référant à la conservation de la nature indépendamment des bénéfices matériels ou de valeurs mesurables et (ii) les valeurs instrumentales ou économiques (Figure 3).

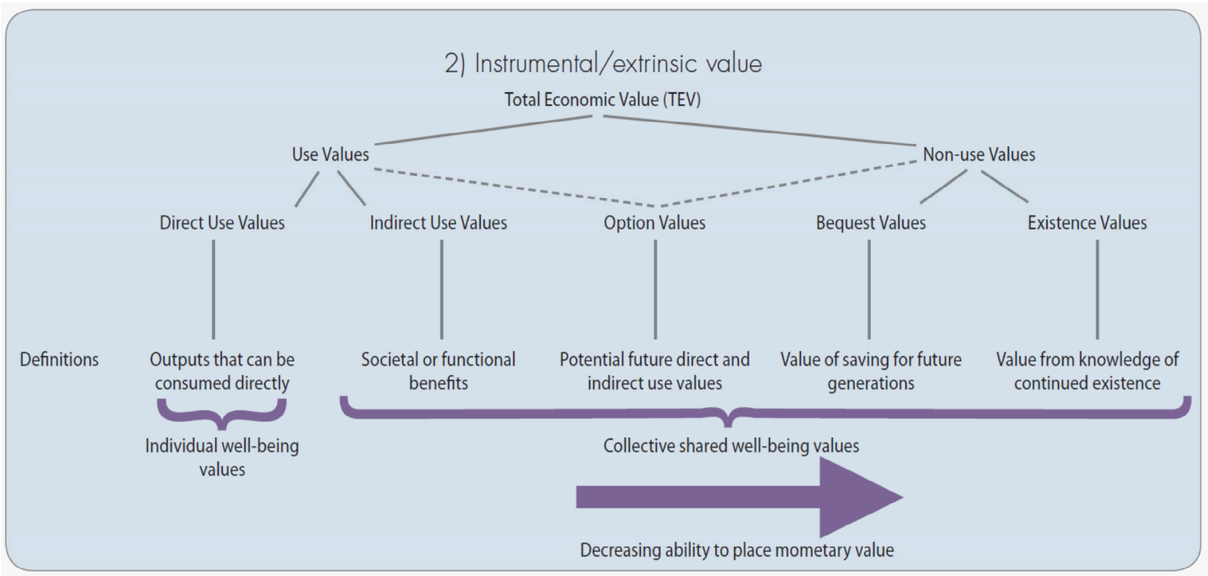


Figure 3 : Les différents types de valeurs économiques attribuées aux services écosystémiques (Mace et al., 2011).

Cette dernière catégorie comprend différentes valeurs selon qu'elles découlent d'une utilisation directe ou indirecte des écosystèmes, ou qu'elles reflètent la satisfaction que les individus dérivent du fait de savoir que les services peuvent être maintenus et sont ou pourront être utilisés par d'autres (Pascual et al., 2010).

Cette évaluation est importante car chaque prise de décision est précédée par la pondération des valeurs entre chaque alternative (Bingham et al., 1995). C'est dans cet esprit que l'initiative TEEB (TEEB 2010, Ring et al. 2010) a souligné l'importance économique de la biodiversité et des services écosystémiques. Cependant, bien que cette approche économique des valeurs associées aux services soit prépondérante, celle-ci peut mener à des résultats discutables tels que la reproduction de logiques économiques et marchandes dans le domaine de la conservation de la nature (Gómez-Baggethun et al., 2010). Ces logiques peuvent avoir des conséquences importantes comme la modification des régimes de propriétés des écosystèmes et le glissement de la conservation par obligations éthiques ou communautaires au profit de l'intérêt financier personnel, ce qui peut mener à des effets opposés à ceux espérés (Gómez-Baggethun et al., 2010). Néanmoins les services écosystémiques peuvent éclairer la prise de décision sans que soit nécessaire une évaluation monétaire (de Groot et al., 2010). En effet, les arbitrages nécessaires à la gestion des écosystèmes et des ressources naturelles peuvent également être formulés sur la base de critères sociaux (Diaz et al., 2011; Quétier et al., 2010b) ou écologiques (compensation écologique).

Une manière de collecter les données nécessaires à la compréhension des ces différents aspects de demande sociale de services écosystémiques est d'impliquer dans les recherches des chercheurs de différentes disciplines mais aussi des participants non académiques tels que des gestionnaires, des acteurs locaux, du public général afin de combiner les savoirs et de rendre la science plus flexible et de répondre plus facilement aux demandes sociétales. Cette combinaison d'approche participative et d'approche interdisciplinaire est appelée transdisciplinarité (Tress et al., 2005). Cette tendance commence à se développer dans les recherches sur les services écosystémiques (Seppelt et al., 2011).

De nombreuses approches ont été développées pour décrire et comprendre les représentations sociales ou les stratégies d'acteurs sociaux mais elles sont rarement confrontées à des données écologiques pour aborder des questions d'environnement (Lowe et al. 2009). Des méthodes d'enquête auprès d'individus, puis une agrégation via des ateliers collectifs (*focus groups)* permettent de révéler les stratégies, priorités et dépendances vis-à-vis de services écologiques (Diaz et al., 2011). Si les entretiens individuels (structurés ou non) offrent l'opportunité d'en approfondir la compréhension ce sont les méthodes collectives qui permettent de comprendre l'influence du contexte social, là où les gens discutent, négocient, hiérarchisent, et reformulent les points de vue, attitudes et comportements des uns et des autres (Kelemen and Gomez-Baggethun, in revision). De telles méthodes participatives ont souvent été considérées dans les études sur la gestion des ressources naturelles (Chevalier and Buckles, 2009; Etienne and (coord), 2010) et ont montré « comment les gens de tous les horizons peuvent prendre part à des analyses complexes de leur propre situation avec toute la compétence voulue » (Chevalier and Buckles, 2009).

2.2 Le système écologique

La biodiversité, définie comme la variété des organismes vivants et des complexes écologiques dont ils font parties, modifie et sous-tend la provision de services des écosystèmes (Chapin et al., 2000; MEA, 2005). Bien que les différentes composantes de la biodiversité, allant de la diversité génétique à la diversité des habitats, peuvent jouer un rôle dans la fourniture d'au moins un service des écosystèmes, certaines composantes ont plus d'influence que d'autres. Un large éventail de recherches sur les relations entre biodiversité et fonctionnement des écosystèmes a mené à la conclusion que c'est plutôt la diversité fonctionnelle (c-à-d. le type, la gamme et plus particulièrement l'abondance des traits fonctionnels) que la diversité spécifique qui régit le fonctionnement des écosystèmes (de Bello et al., 2010; Diaz et al., 2006; Hooper et al., 2005). De plus, il a également été démontré que plusieurs traits individuels peuvent simultanément changer la fourniture de multiples services, et qu'un service donné dépend souvent de multiples traits (de Bello et al., 2010; Lavorel and Grigulis, 2012).

Un trait est une caractéristique morphologique, écophysiologique ou génétique d'un organisme (plante ou espèce animale) qui a un lien démontré avec la fonction de l'organisme (Lavorel et al., 1997). La réponse d'une plante à son environnement et son effet sur les communautés d'espèces végétales et animales ou microbiennes, ou sur l'écosystème, peut être prédit par ses traits (Figure 4) (Lavorel & Garnier 2002). Ce trait est dit « fonctionnel », car il possède une fonction pour la plante et lui permet de subsister dans son environnement. Ces traits peuvent concerner la plante entière (ex. la hauteur végétative), les feuilles (ex. la taille, le contenu en matière sèche, la concentration en azote ou phosphore), les parties souterraines (ex. le diamètre des racines) ou les parties reproductives (ex. la masse de graine) (Cornelissen et al., 2003). L'hypothèse centrale est que des espèces ayant des traits similaires vont avoir une même réponse (trait de réponse) à un facteur de l'environnement et/ou un même effet sur l'écosystème (trait d'effet). Les influences des facteurs environnementaux sur les organismes déterminent des traits de réponse mesurables, qui à leur tour ont donc des effets sur les services écosystémiques (Lavorel and Garnier, 2002). Les traits fonctionnels peuvent être utilisés à deux niveaux d'agrégation différents : l'espèce et la communauté. A l'échelle de la communauté, on calcule la moyenne d'un trait pondérée par la biomasse de chacune des espèces qui composent la communauté (Lavorel and Garnier, 2002).

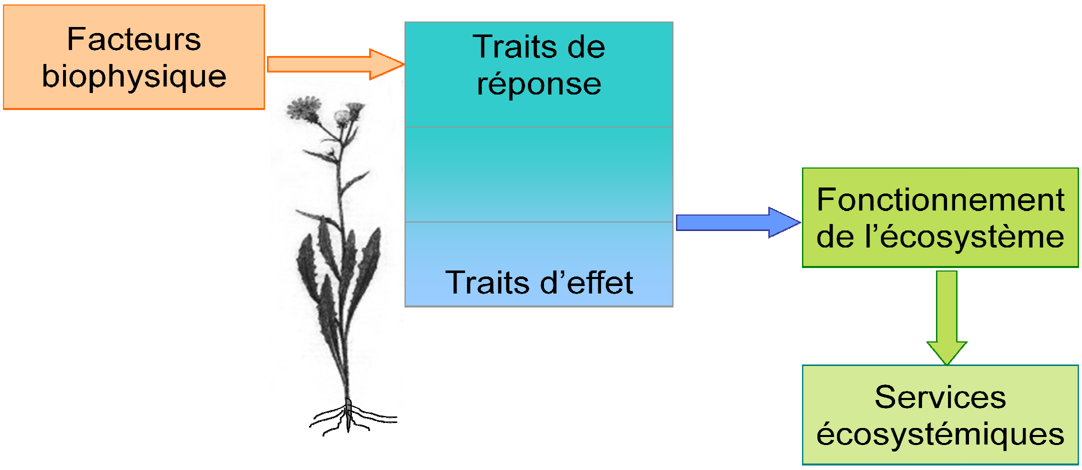


Figure 4 : Relations entre facteurs biophysiques (ex. altitude, topographie, climat, sols) et structure des communautés ou fonctionnement des écosystèmes. Les traits fonctionnels sont considérés comme centraux pour prévoir la réponse des écosystèmes face aux changements des facteurs et leur fonctionnement futur (adapté de Lavorel & Garnier 2002).

Sur la base de ces connaissances, il a été proposé que des approches basées sur les traits fonctionnels puissent fournir un cadre pour identifier les mécanismes sous-tendant la fourniture de services des écosystèmes (Diaz et al., 2007) et constituer un outil pour prédire les changements de services qui résultent des impacts de la gestion sur les plantes et la biodiversité du sol (Quétier et al., 2007a). Par exemple, dans le cas des prairies, l'usage d'une telle approche qui reconnait que de nombreux services écosystémiques sont sous-tendus par des traits végétaux identiques et par l'effet combiné et l'interaction des traits végétaux et microbiens (de Bello et al., 2010), peut donc aider à élucider les mécanismes qui contrôlent la fourniture de multiples services.

L'effet direct des traits végétaux sur les écosystèmes est bien connu. Mais, ils influencent aussi indirectement les processus écosystémiques par leur action sur les communautés microbiennes du sol et sur les cycles du carbone et des nutriments. Par exemple, des traits végétaux qui influencent la qualité de la litière végétale, comme le taux de croissance, la taille, la longévité (Wardle et al., 1998), la surface foliaire spécifique (Kazakou et al., 2006), ou le contenu en matière sèche des feuilles, peuvent tous potentiellement altérer les communautés biologiques du sol et le taux de décomposition de la litière, et par conséquent les cycles du carbone et de l'azote (de Bello et al., 2010; De Deyn et al., 2008). De même, des traits souterrains (ex. absorption de l'azote, exsudats racinaires) peuvent fortement influencer l'activité, la structure et la diversité des communautés microbiennes de la rhizosphère (Schweitzer et al., 2008), ce qui en retour peut altérer les cycles du carbone et de l'azote et les relations avec les communautés végétales (Wardle et al., 2004).

2.3 A l'interface des sous-systèmes : la gestion des terres et les services écosystémiques

La connaissance de l'effet de différents types gestion des terres sur les services écosystémiques est essentielle pour prendre des décisions adéquates et durables dans le temps. En effet, différentes utilisations des terres peuvent conduire à la fourniture de différents groupes de services (*bundles*) (Raudsepp-Hearne et al., 2010) suite aux mécanismes biophysiques qui sous-tendent leur fourniture. Par ailleurs, des compromis (*trade-offs*) entre services peuvent apparaitre quand la fourniture d'un service est réduite du fait de l'augmentation d'un autre service (Rodriguez et al., 2006). Ceux-ci peuvent avoir lieu dans l'espace lorsque la décision d'augmenter un service à un endroit a un effet sur un service à un autre endroit ou à une échelle plus large (Hein et al., 2006). Ces compromis peuvent aussi avoir une dimension temporelle car les décisions de gestion des terres se focalisent généralement sur la provision immédiate d'un service qui peut avoir un effet négatif sur la fourniture d'autres services dans le futur. L'analyse de tels compromis requière des informations supplémentaires sur les interactions entre services et leur compatibilité (Lavorel and Grigulis, 2012; Raudsepp-Hearne et al., 2010). Par ailleurs, des arbitrages peuvent également avoir lieu suite à la localisation de besoins et d'attentes hétérogènes de la part d'acteurs sociaux concernés (Diaz et al., 2011)**.**

Les deux sous-sections suivantes abordent, respectivement, ces aspects spatiaux et temporels propres aux services écosystémiques.

2.3.1 Nécessité d'une spatialisation des services

La cartographie des services écosystémiques représente un outil important pour les décideurs politiques et les institutions car il permet d'évaluer les relations spatiales (compromis et synergies) entre services, ainsi que d'identifier les zones qui sont susceptibles de fournir le plus de services répondant à différents objectifs de gestion ('hot spots') (Anderson et al., 2009; Egoh et al., 2008; Nelson et al., 2009; Raudsepp-Hearne et al., 2010). C'est pourquoi un nombre croissant d'études cartographie la distribution spatiale des services des écosystèmes, et ce principalement à l'échelle régionale. Bien qu'il soit reconnu que les mesures, la modélisation et le suivi des processus et fonctions des écosystèmes sont un pré-requis pour l'évaluation des services des écosystèmes (Carpenter et al., 2009; Rounsevell et al., 2012), l'approche la plus commune pour cartographier les services repose sur l'utilisation de variables de substitution (*proxy*) (Martinez-Harms and Balvanera, 2012; Seppelt et al., 2011), particulièrement l'occupation et l'utilisation du sol, et de données topographiques (ex. Egoh et al., 2008; Nelson et al., 2009; Swetnam et al., 2011) qui sont relativement faciles à acquérir. Pourtant Eigenbrod et al. (2010) ont démontré que les cartographies de services prenant en compte uniquement l'occupation du sol ou se basant sur des *proxy* conduisent à un mauvais ajustement avec celles basées sur des données primaires, notamment pour des études à échelles locales, car la variabilité spatiale des variables biophysiques ou des processus ne sont pas pris en compte. Les autres méthodes de cartographie reposent sur l'extrapolation de données mesurées à une échelle plus fine (Raudsepp-Hearne et al., 2010) notamment par l'utilisation de modèles statistiques (Lavorel et al., 2011) ou plus rarement sur les connaissances d'experts (Egoh et al., 2008) (pour plus de détails voir Martinez-Harms and Balvanera, 2012). Le fait que la plupart des modélisations spatiales de services font appel à des données secondaires (tableaux de références) et à des techniques de modélisation simples résulte de contraintes d'accès aux données, de temps et de budget disponible (Martinez-Harms and Balvanera, 2012). Toutefois, malgré le besoin d'une communication simple et efficace avec les décideurs politiques, une simplification exagérée des modèles n'est pas satisfaisante pour la description d'un système global et peut induire en erreur les prises de décisions (Seppelt et al., 2011). Généralement, les cartographies de services se focalisent sur la fourniture potentielle de services (fonctions) sans tenir compte de la localisation de la demande (Petz and van Oudenhoven, 2012).

Actuellement, cette thématique de recherche souffre du manque, d'une part, de connaissance des dynamiques écologiques et de l'effet de différentes modalités de gestion sur la fourniture potentielle de services et, d'autre part, d'indicateurs facilement mesurables et/ou modélisables des besoins et attentes sociétales (Rounsevell et al., 2012; Strager and Rosenberger, 2006).

2.3.2 Nécessités d'une démarche prospective

Les choix de gestion des écosystèmes impliquent des arbitrages entre services écosystémiques non seulement dans l'espace mais aussi dans le temps. Ces choix impliquent parfois de privilégier un service aujourd'hui (ex. un service d'approvisionnement comme la production agricole) au dépens d'un autre service à plus long terme (ex. un service de régulation comme la qualité de l'eau). Essayer de se rapprocher de situation « gagnant-gagnant » demande de comprendre où, quand, comment, pourquoi et quels services seront affectés par des changements (ex. utilisation du sol). Cependant, l'analyse de ces dynamiques temporelles des services écologiques n'est que rarement prise en compte. Celle-ci nécessite qu'on « situe » les écosystèmes concernés sur une ou des trajectoires écologiques le long desquelles ils pourraient être pilotés. Quétier et al (2007b) ont ainsi montré l'intérêt de considérer non seulement les différences de gestion actuelle de prairies, mais aussi leurs historiques afin d'étudier la fourniture de services à une échelle locale. Une autre étude a examiné comment l'évolution historique de l'occupation et l'utilisation du sol à une échelle régionale a affecté la fourniture de services (Lautenbach et al., 2011).

L'évaluation des avantages et inconvénients futurs de chaque option de gestion des terres envisagée fait nécessairement appel à une démarche de projection concernant les fonctions écosystémiques (l'offre) et/ou les besoins et attentes à satisfaire (la demande). Les scénarios prospectifs constituent un outil efficace pour évaluer et comprendre les cheminements possibles de changement de fourniture de services (Peterson et al., 2003). Les scénarios sont des descriptions plausibles et souvent simplifiées de la façon dont le futur pourrait se développer sur la base d'hypothèses cohérentes sur les principaux facteurs de changement et leur interrelations (tel que défini par Millennium Ecosystem Assessment, 2005). L'incertitude des projections augmentant à mesure qu'on se projette dans le long terme, il peut être utile d'évaluer les changements de services écosytémiques selon différents contextes prenant en compte plus ou moins de variation des facteurs causaux (drivers) tel que le climat, les marchés et les politiques, et les effets d'impacts cumulés. Cependant, une minorité d'études prennent en compte l'effet de scénarios sur l'évolution des services des écosystèmes. Celles-ci sont en effet souvent focalisées sur des options politiques ou des changements comportementaux, plus que sur des effets climatiques ou des effets combinés (Seppelt et al., 2011).

2.4 Effets de rétroaction du changement de services sur la décision des *stakeholders*

Si les changements d'utilisation du sol et de services écosystémiques associés peuvent être causés par des changements socio-économiques exogènes au système socio-écologique (ex. urbanisation, développement économique), ceux-ci peuvent aussi être régis par des processus endogènes au système (Lambin and Meyfroidt, 2010). En effet, des changements de décisions de gestion peuvent résulter d'une dégradation sévère des services écosystémiques causés par des pratiques d'utilisation du sol passées (Lambin and Meyfroidt, 2010). Cet effet de rétroaction peut-être négatif dans le cas d'un déclin ou d'un épuisement des ressources naturelles qui par des enchainements en cascade va altérer les dynamiques sociales, ce qui en retour peut accentuer la dégradation des écosystèmes (Carpenter et al., 2006). Il a en outre été montré que ces effets de rétroaction endogène influencent davantage les décisions à l'échelle locale qu'à l'échelle nationale (Lambin and Meyfroidt, 2010). Par conséquent, comprendre quels sont les facteurs qui déterminent ces choix et comment ces décisions sont prises par les sociétés et les individus est important. Meyfroidt (2012) suggère que les décisions de gestion des terres sont issues d'arbitrages basés sur la façon dont les services et les changements sont perçus, interprétés et évalués par les *stakeholders*. Les effets de rétroactions du changement de services sur la décision peuvent être soit directs c'est-à-dire affectant seulement les paramètres de décisions, soit indirects, affectant les connaissances, les valeurs et les stratégies qui sous-tendent les décisions (Meyfroidt, in revision). Pourtant, il est rare que des études portant sur la perception des ressources naturelles les relient aux comportement des individus (Meyfroidt, 2012), et il n'y a à ma connaissance pas d'étude traitant spécifiquement cet aspect en ce qui concerne les services écosystémiques.

Afin d'analyser l'effet des services sur les arbitrages visant à satisfaire une ou plusieurs composantes du bien-être humain, Turner (2010) suggère de combiner des approches adaptées d'une part, à l'étude de la stabilité des écosystèmes face aux perturbations et, d'autre part, à la capacité d'adaptions des *stakeholders*.

3 Objectifs et structure de la thèse

3.1 Objectifs et cadre théorique

Le concept de services écosystémiques amène à étudier les liens complexes entre l'homme et son environnement, en mettant l'accent sur les bénéfices qu'il retire des écosystèmes. Malgré le nombre croissant de cas d'étude considérant différentes dimensions des interactions entre les écosystèmes et l'utilisation du sol via les services écosystémiques, les recherches transdisciplinaires sur les relations entre la biodiversité, les services et l'utilisation du sol par une approche socio-écologique restent théoriques. Cette thèse a donc pour objectif principal de combler ce manque en explorant et analysant les dynamiques et processus des services écosystémiques, y compris les effets de rétroactions, par une approche socio-écologique dans un contexte de changement planétaire (utilisation du sol, climat, économie, politique) (Figure 2). A cette fin, j'appliquerai ce cadre conceptuel (Figure 5) à l'analyse d'un site d'étude de prairies de montagne, en insistant sur l'effet de rétroaction entre les services écosystémiques et la prise de décision des acteurs en terme de gestion des écosystèmes. En effet, dans ces écosystèmes, la fourniture potentielle des services écosystémiques dépend fortement de l'état de l'écosystème, lui-même directement affecté par la gestion humaine.

L'objectif général de la thèse peut être décliné en cinq questions plus précises :

1. Quels services écosystémiques sont perçus, utilisés et/ou appréciés par les *stakeholders* ? (boite 3 de la Figure 5)
2. Quels services écosystémiques sont potentiellement fournis par les prairies étant données les dynamiques écologiques ? (relations entre les boites 1, 2 et 6 de la Figure 5)
3. Comment la gestion d'utilisation du sol affecte la fourniture de services écosystémiques ? (relation entre les boites 4 et 1 de la Figure 5)
4. Comment les services écosystémiques sont pris en compte dans les processus de décisions de gestion d'utilisation du sol, sous l'angle de l'effet de rétroaction des services sur l'utilisation du sol ? (boite 4 la Figure 5)
5. Comment le contexte global influence ce système socio-écologique ? (effets des boites 5 et 6 sur les autres boites de la Figure 5 dans le cadre de différents scénarios)

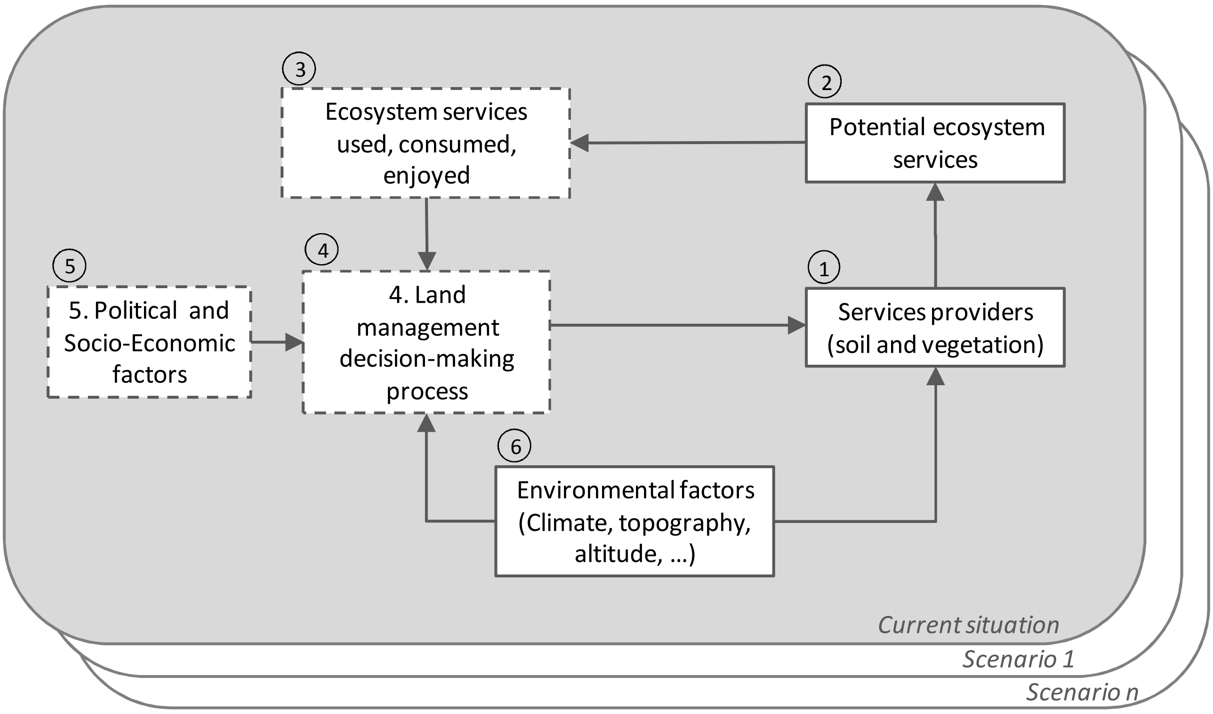


Figure 5 : Cadre conceptuel de la thèse. Les boites tiretées représentent le sous-système socio-économique et les boites en trait plein représentent le sous-système écologique. Ce système peut être étudié sous différents contextes de changement globaux dans le cadre de scénarios prospectifs.

Au-delà de l'objectif d'amélioration des connaissances scientifiques sur les services écosystémiques fournis par un socio-écosystème montagnard, cette thèse a aussi un objectif méthodologique. J'ai à cet égard mis en œuvre une approche transdisciplinaire novatrice combinant des données qualitatives et semi-quantitatives recueillies par des méthodes participatives et l'utilisation de données écologiques, afin d'explorer et analyser les liens entre les différentes composantes du socio-écosystème, y compris l'effet de rétroaction du changement de niveau de services sur la prise de décision des agriculteurs. Ce cadre conceptuel et méthodologique est mis à l'épreuve grâce au cas d'étude des prairies subalpines de la commune de Villar d'Arène (Hautes-Alpes, France).

J'ai mené l'ensemble de ces travaux en collaboration avec des spécialistes des différentes disciplines (agronomie, sociologie, géographie, écologie) mobilisées dans chacune des questions étudiées.

3.2 Structure de la thèse

Les résultats de ce travail de thèse sont présentés en trois parties correspondant chacune à l'étude empirique d'une composante du socio-écosystème de Villar d'Arène (présenté à la fin de ce chapitre). La première partie explore les relations entre les écosystèmes et les services écosystémiques (chapitres 2 à 4). La deuxième partie étudie l'influence de l'utilisation du sol sur la fourniture de services écosystémiques dans différents contextes climatiques et socio-économiques (chapitres 5 et 6). La troisième partie « ferme la boucle » du système socio-écologique en explorant l'effet de rétroaction du changement de services sur l'utilisation du sol au travers des processus de décision des agriculteurs (chapitre 7). Finalement, la discussion synthétise les principaux résultats de la thèse et évalue leurs apports sur le plan de la compréhension des services écosystémiques.

Chapitre 2. Ce chapitre passe en revue la diversité de l'utilisation du concept de services écosystémiques dans le monde académique et non-académique et précise sous quel angle les services des écosystèmes sont abordés dans notre travail.

Il est l'objet d'un article publié dans Comptes Rendus Biologies : Lamarque, P., Quetier, F., Lavorel, S., 2011, The diversity of the ecosystem services concept and its implications for their assessment and management, *Comptes Rendus Biologies* 334(5-6):441-449

Chapitre 3. Ce chapitre définit les services écosystémiques fournis par les prairies de montagne en explorant l'importance attribuée aux services perçus par différents types de *stakeholders*, ainsi que leurs connaissances sur ce sujet.

Il est l'objet d'un article publié dans *Regional Environmental Change* : Lamarque, P., Tappeiner, U., Turner, C., Steinbacher, M., Bardgett, R. D., Szukics, U., Schermer, M., Lavorel, S., 2011, Stakeholder perceptions of grassland ecosystem services in relation to knowledge on soil fertility and biodiversity, *Regional Environmental Change*, 11(4):791-804.

Chapitre 4. Ce chapitre propose et met en œuvre sur notre zone d'étude une méthode de cartographie des services écosystémiques basée sur les traits fonctionnels.

Il est l'objet d'un article publié dans *Journal of Ecology* : Lavorel, S., Grigulis, K., Lamarque, P., Colace, M.-P., Garden, D., Girel, J., Pellet, G., Douzet, R., 2011, Using plant functional traits to understand the landscape distribution of multiple ecosystem services, *Journal of Ecology* 99(1):135-147.

Chapitre 5. Ce chapitre propose une méthode participative de construction de scénarios prospectifs et l'applique au développement de scénarios climatiques et socio-économiques pertinents à l'échelle régionale, ainsi que des scénarios d'utilisation du sol dans ces différents contextes.

Il est l'objet d'une publication soumise à *Landscape and Urban planning* : Lamarque, P., Nettier, B., Barnaud, C., Artaux, A., Eveilleau, C., Dobremez, L., Lavorel, S., submitted, A participatory approach to map land management change based on the adaptive management of mountain livestock systems to drought and socio-economic scenarios, *Landscape and Urban Planning*.

Chapitre 6. Ce chapitre évalue l'effet des changements climatiques et d'utilisation des terres associés à chacun des scénarios sur la fourniture de services écosystémiques, pris individuellement ou selon leurs interrelations.

Il est l'objet d'une publication en préparation pour PNAS : Lamarque, P. Lavorel, S., Quétier, F., Mouchet M., Direct and indirect effects of climate change on bundles of grassland ecosystem services.

Chapitre 7. Ce chapitre explore l'effet de rétroaction du changement de services sur l'utilisation du sol au travers des processus de décision des agriculteurs.

Il est l'objet d'une publication en préparation pour Ecology and Society : Lamarque P. Meyfroidt P. Barnaud C., Nettier B., Lavorel S. , Ecosystem services in French mountain farmers' decision-making.

Les chapitres centraux (2 à 7) rédigés sous forme d'article pour des revues internationales sont présentés conformément à la version intégrale de l'article auquel ils sont associés. Ceux-ci peuvent être lus séparément, ce qui occasionne quelques redondances entre chapitres. A la fin de chaque partie, une synthèse en français fait les liens entre les principaux résultats et présente éventuellement quelques résultats non publiés.

4 Le socio-écosystème de Villar d'Arène

4.1 L'agriculture de montagne

Les régions de montagne sont caractérisées par des systèmes biophysique et socio-économique fragiles et marginaux (Beniston, 2000). En Europe centrale, ce sont les Alpes qui possèdent les plus grandes surfaces d'écosystèmes naturels et semi-naturels, mais ceux-ci sont menacés par des pressions croissantes provenant de changements d'activités agricoles et touristiques ainsi que du changement climatique (Pauli et al., 2003; Tasser and Tappeiner, 2002). Les prairies représentent la majorité des écosystèmes alpins et sont principalement utilisées comme pâturages et surfaces de fauche (les prés constituent 80% de la surface agricole utile (SAU)) en raison des marchés agricoles européens, du contexte économique et des conditions naturelles (Tappeiner et al., 2008a). Les prairies permanentes sont de plus en plus reconnues pour les multiples services écosystémiques qu'elles fournissent, tels que la conservation de la biodiversité, la régulation des flux physiques et chimiques (Gibon, 2005a; Lemaire et al., 2005). C'est pourquoi, dans les contextes politique et socio-économique actuels, la multifonctionnalité représente un atout potentiel pour maintenir ces systèmes fragiles.

Malgré ces atouts, durant la dernière décennie, les influences socio-économiques, politiques et technologiques ont amené l'agriculture de montagne à se concentrer sur les terrains les plus accessibles et les plus fertiles au détriment des autres (MacDonald et al., 2000) menant à des changements d'utilisation du sol, de paysage et de biodiversité (Maurer et al., 2006; Soliva, 2007). En réponse aux préoccupations des gestionnaires et décideurs politiques, ces changements ont attiré l'attention de nombreux scientifiques sur les agro-écosystèmes montagnards (Aarnink et al., 1999; CIPRA Internationale Alpenschutzkommission, 2001; Tappeiner, 2003). Par ailleurs, comme les agro-écosystèmes sont les écosystèmes les plus directement gérés par l'homme pour répondre à ses besoins, ceux-ci jouent un rôle capital à la fois sur l'offre et la demande de services écosystémiques et la décisions des acteurs (Swinton et al., 2007). Par conséquent, comprendre comment les actions humaines et les autres facteurs affectent la fourniture de services écosystémiques dans le cas des agro-écosystèmes montagnards semble une priorité.

C'est pourquoi j'ai choisi d'étudier les services écosystémiques dans un agro-écosystème de prairies subalpines dans les Alpes françaises.

4.2 La Zone Atelier Alpes

Les Zones Ateliers (ZA) sont des dispositifs de recherche dont la vocation est de mieux comprendre les relations entre une société et son environnement. Au niveau national, les objectifs généraux des ZA sont de : (1) favoriser le dialogue entre sciences sociales et sciences de la nature ; (2) développer des recherches sur un grand territoire présentant une unité fonctionnelle ; (3) d'étudier les interactions entre écosystèmes et dynamiques économiques, socio-politiques, techniques ; (4) prendre en compte le long terme (passé et futur). Au niveau international, les ZA sont intégrées dans le réseau d'excellence européen Alter-Net et le réseau des plates-formes de recherche socio-écologiques à long terme (Long-Term Socio-Ecological Research (LTSER)) qui se concentrent sur les interactions entre système social et système naturel en combinant des méthodes et approches interdisciplinaires (Haberl et al., 2006). Créée en 2008, la « Zone Atelier Alpes » rassemble des chercheurs en écologie, en mesures physiques et en sciences sociales ainsi que plusieurs gestionnaires d'espaces naturels. Ses travaux portent sur l'évolution des écosystèmes et du climat dans des territoires marqués par l'importance de certaines activités humaines, en particulier pastorales et touristiques (Lavorel et al., 2012). Elle se concentre sur deux sites alpins français (Figure 6) contrastés par leurs conditions naturelles et humaines : (1) le Vercors (Hauts-Plateaux), (2) l'Oisans (Huez et Lautaret) dont la station principale se situe au col du Lautaret sur la commune de Villar d'Arène. Sur cette commune, des données sur la végétation, le climat et les systèmes agricoles sont collectées de manière approfondies par le Laboratoire d'Ecologie Alpine depuis 2003.

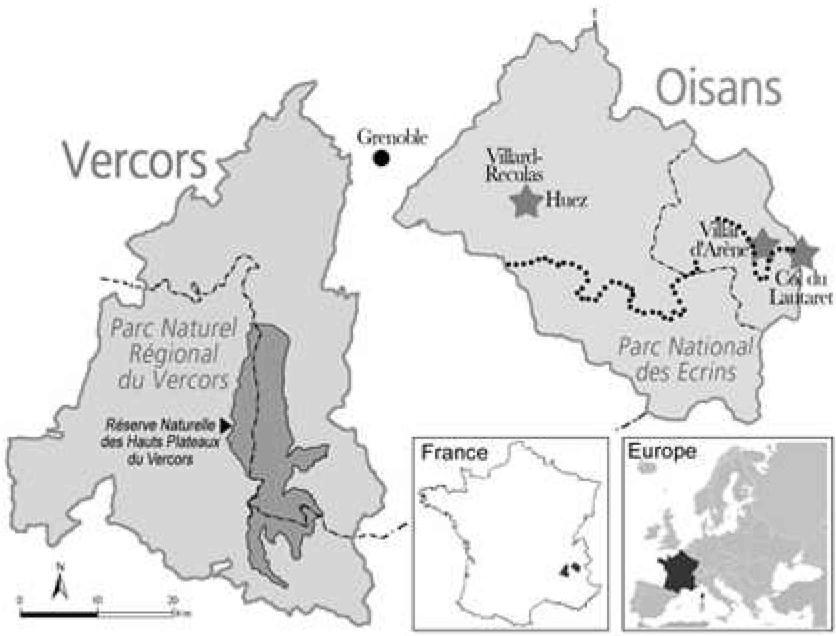


Figure 6: Sites de la Zone Atelier Alpes (http://www.za-alpes.org/).

4.3 Villar d'Arène (Col du Lautaret)

Dans ce contexte, Villar d'Arène a été retenu comme site d'étude pour: (1) la possibilité d'étudier la fourniture et la demande de nombreux services susceptibles d'être menacés à moyen terme par des changements de gestion des terres, (2) la disponibilité par la zone atelier de nombreuses connaissances et données écologiques, agronomiques et socio-économiques requises pour mener à bien une approche socio-écologique dans sa globalité.

4.3.1 Description générale

Villar d'Arène (45°02'N, 6°20 E) est une commune du Canton de La Grave situé dans le département des Hautes-Alpes (Région Provence-Alpes-Côte d'Azur) à la limite du département de l'Isère (Région Rhône-Alpes). La commune est composée de 4 hameaux (Le village, Les Cours, Le Pied du Col et le Col du Lautaret) au sein desquels sont répartis 284 habitants (INSEE,2008) appelés les Faranchins. Son adret majoritairement composé de prairies fait partie de la zone d'adhésion du Parc Naturel des Ecrins, alors que son ubac majoritairement composé de forêts de mélèze, de roche et de glaciers fait partie de la zone cœur du parc (Figure 7). Cette étude se concentrera donc sur la partie adret (Figure 8) dont la surface totale compte 1300 hectares étagés entre 1550 et 2500m d'altitude.

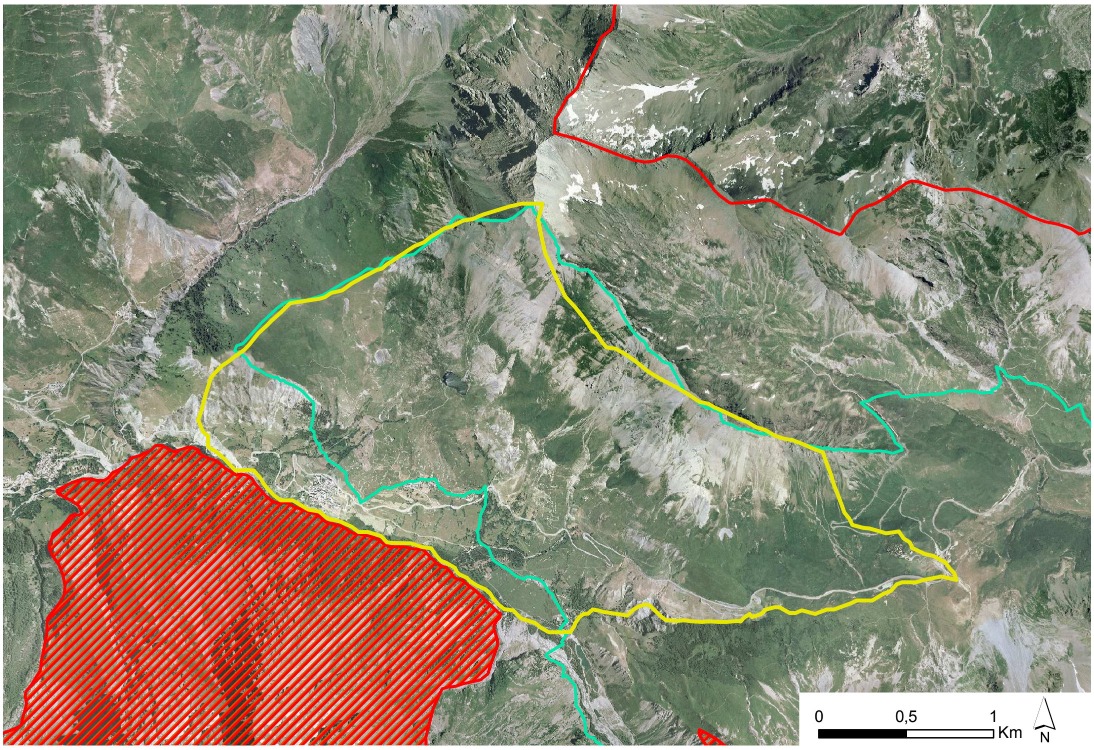


Figure 7 : Site d'étude (contour jaune) et ses zonages administratifs : cœur du parc (rouge hachuré) et zone d'adhésion (contour rouge), zone N2000 Combeynot-Lautaret-Ecrins (contour vert). Au nord, la zone adret majoritairement occupées par des prés de fauches, des pâturages et des alpages. Fond de carte © BD Ortho IGN, 2003.

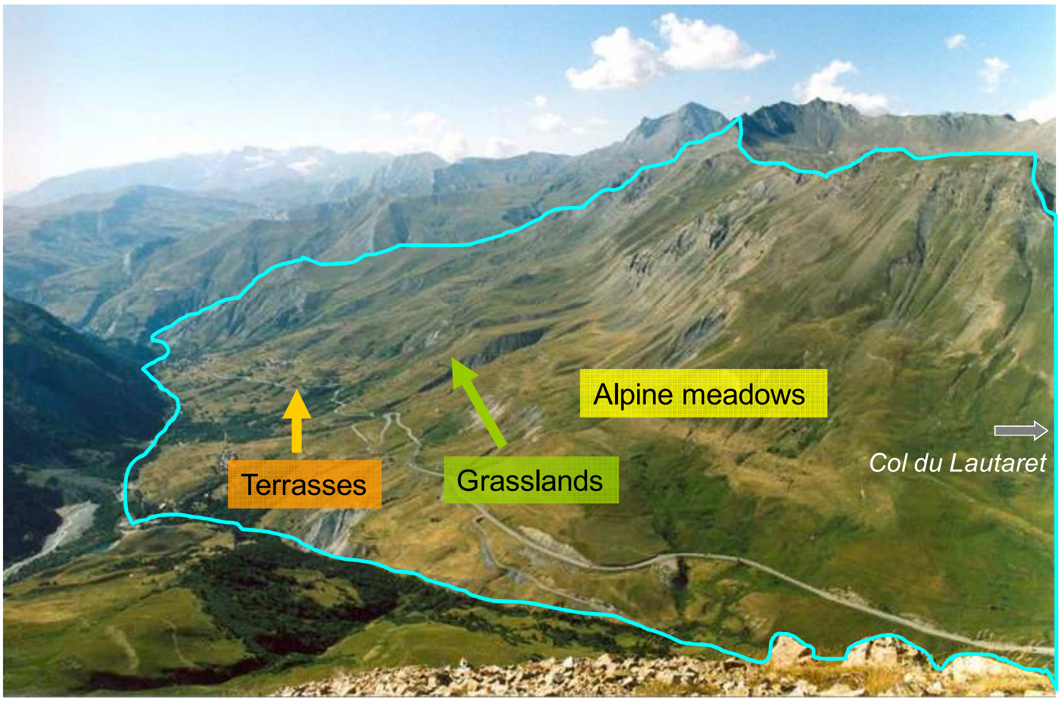


Figure 8 : Adret de Villar d'Arène vu depuis le Laurichard (Photo F. Quétier, 2003) et la localisation des terrasses, prairies en dehors des terrasses et alpages.

Située dans une zone intermédiaire entre les Alpes internes et les Alpes externes, cette vallée protégée des précipitations par les massifs externes (Ozenda, 1985), connait une pluviométrie annuelle moyenne relativement faible de 960mm, dont 18% en été (Quétier, 2006). Les amplitudes thermiques sont importantes (-7,4°C en février et 19,5°C en juillet en moyenne à proximité du col du Lautaret) (Bakker et al., 2008)). Ces facteurs, couplés à une fonte des neiges parfois tardive (fin avril voir mai) réduit la saison de végétation à 6 mois en moyenne (Robson et al., 2010). Le haut de pente est constitué d'un sol peu profond sur un substrat homogène de schistes argileux. Le bas des pentes sont quant à eux essentiellement composés de dépôts glaciaires comportant quelques éléments de schiste calcaire (Bakker et al., 2008). Ces particularités expliquent l'origine d'une diversité floristique spécifique importante au niveau régional (Quétier, 2006). Toutefois, ces prairies subalpines ne représentent pas la végétation climacique qui correspond pour ce site à une forêt de *Pinus cembra* (pin cembro (Ozenda, 1985)). L'absence de forêt à cette altitude correspond à une modification progressive anthropique du milieu liée à une utilisation agricole depuis l'Age du Bronze (Girel et al., 2010), avec une accentuation à partir du 8ème/9ème siècle des déboisements intensifs dans le but d'augmenter la surface de pâture d'altitude. Toutefois, plusieurs phases de déforestation et de recolonisation ont vu le jour au cours des trois derniers millénaires en fonction des contextes économiques, politiques et épidémiques (Girel et al., 2010).

Dans la suite de cette partie, je présente une description des éléments du système socio-écologique de Villar d'Arène, en commençant par le contexte agricole passé et actuel ainsi que la végétation des différents types de prairies. Ensuite, le tourisme première activité économique du canton est brièvement présenté et, pour finir, nous présenterons les aspects des politiques publiques importants pour l'agriculture et la conservation de la biodiversité sur le site.

4.3.2 Contexte historique de l'agriculture faranchine[2]

Le paysage de Villar d'Arène a très tôt (optimum démographique médiéval à la fin du 14ème siècle) été caractérisé par l'établissement de terrasses sur les parties les plus basses de l'adret (Figure 9). Ce système de terrasses est courant dans les régions du monde où les labours doivent être installés sur des zones pentues mais les terrasses de l'Oisans ont la particularité d'être séparées par des talus plutôt que par des murets de pierre. Elles ne sont dès lors pas horizontales, mais possèdent une pente inférieure à la pente initiale. Ces terrasses étaient principalement vouées à la production de céréales (orge, seigle, avoine, froment) gérées par un assolement triennal, et de pomme de terre (après 1750). Les champs de céréales les plus élevées pouvaient atteindre 2000 mètres d'altitude.

[2] La première partie traitant de l'histoire de l'agriculture est basée sur l'ouvrage de Girel, J., Quétier, F., Bignon, A., Aubert, S., 2010, Histoire de l'agriculture en Oisans. Hautes Romanche et pays faranchin. Villar d'Arène, Hautes-Alpes, *La Galerie de l'Alpe*, Station Alpine Joseph Fourier, Grenoble, France, pp. 79.

Figure 9: Photographie représentant les terrasses anciennement cultivées autour du hameau des Cours (Photo : E. Deboeuf, août 2009)

Vers 1830, la population atteint son optimum (environ 500 personnes à Villar d'Arène) ce qui conduit les communautés rurales à produire des ressources en grandes quantités. L'usage important de fumier en tant que fertilisant nécessaire à un niveau de production acceptable et comme combustible de chauffage suite au manque de bois, explique l'importance du bétail (moutons, vaches, chèvres, mulets) et du pâturage, ainsi que l'utilisation de toutes les sources de fourrage (herbe, paille, feuille) ou de litière (fougère, feuilles mortes, …). Ce système agricole induisait une répartition spatiale particulière des pratiques sur le versant. Les champs de céréales étaient situés autour des villages aux altitudes les plus basses, alors que la fauche était réalisée jusqu'à 2400m d'altitude à une heure de marche des habitations. Le foin était stocké dans des granges situées à différentes altitudes. Les troupeaux restaient sur les communaux sans revenir chaque soir dans les étables afin d'assurer l'auto fertilisation des pâtures d'altitude. La vaine pâture assurait le passage de l'ensemble des animaux sur les jachères, ce qui permettait une répartition égale des fertilisants. Afin de préserver et utiliser les ressources fourragères à bon escient, des règlements très précis fixaient le nombre de bêtes, la répartition du type de bétail sur la commune. Dès 1836, 1/6ème des labours étaient reconvertis en prairies en raison de l'évolution de l'agro-écosystème vers l'élevage et la culture de l'herbe. En 1852, un transhumant ovin utilise pour la première fois une partie des alpages communaux. En 1885, la mise en place d'une fruitière pour la fabrication de fromage à Villar d'Arène va accélérer le développement d'un élevage laitier et des prairies de fauches associées, aux dépens des cultures vivrières. La modernisation des voies de communication favorisant les échanges entre la plaine et la montagne a mené à l'arrêt du labour, d'abord sur les terrasses les plus éloignées puis sur la quasi-totalité des terrasses dans les années 60 (Figure 10). La fauche va alors se concentrer sur les prairies de fauche de hautes altitudes et progressivement sur les terrasses avec l'abandon du labour (Figures 10, 11, 12). C'est ensuite la mécanisation avec l'apparition de la motofaucheuse en 1950 et des tracteurs dans les années 60-70 qui conduira finalement à abandonner la fauche en faveur de la pâture sur les terrains les plus difficiles d'accès, les plus petites et les plus pentus. Avec l'exode rural, le ramassage du lait et la production de fromage ont disparu dans les années 70. Sous l'influence de différents facteurs, l'agriculture de Villar a donc changé, passant de l'autarcie alimentaire à la spécialisation agro-pastorale (Girel et al., 2010).

a) b)

Figure 10 : a) Labour des terrasses maintenant fauchée et b) fauche vers l'Aiguillon en 1953 sur des secteurs maintenant dédiés à l'alpage. (MuCEM, Marcel Maget)

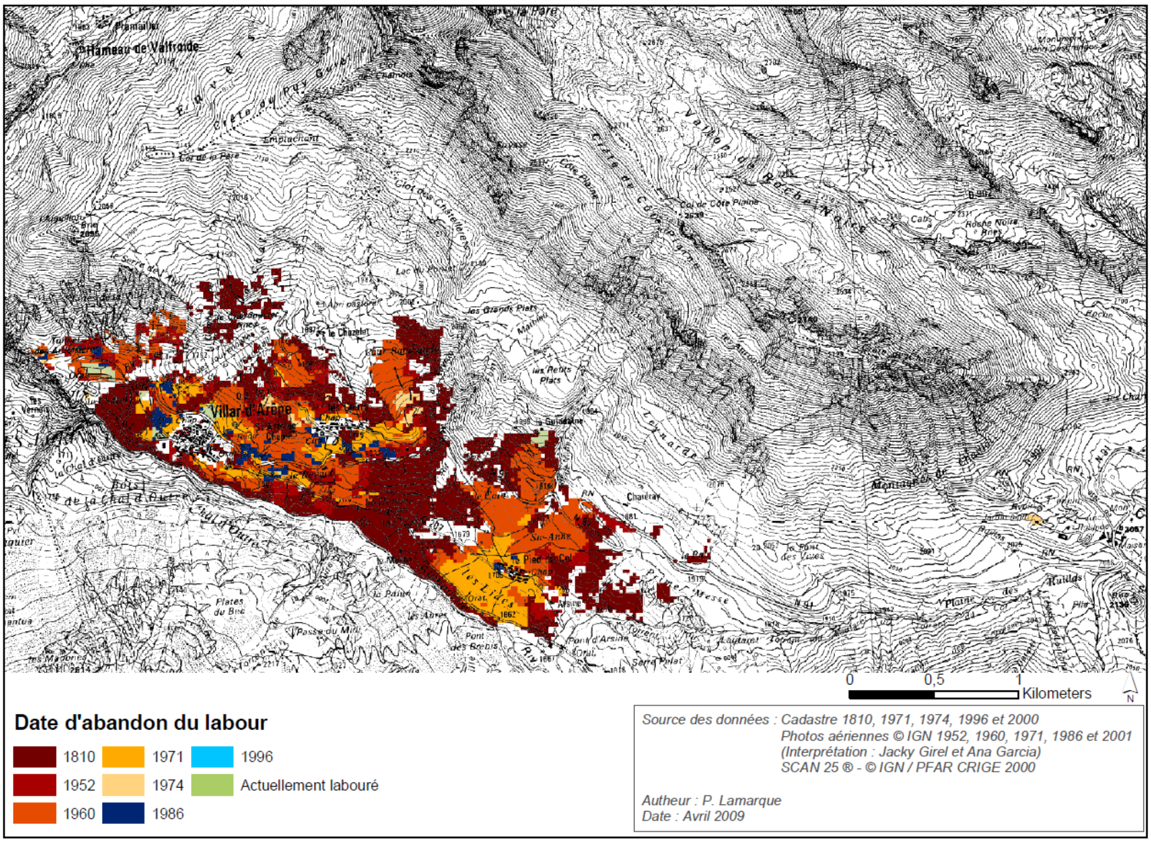


Figure 11: Dates d'abandon du labour à partir de l'interprétation des photos aériennes de 1952, 1960, 1971, 1974, 1986, 1996 et 2001, et des plans cadastraux de 1810, 1971 et 2001.

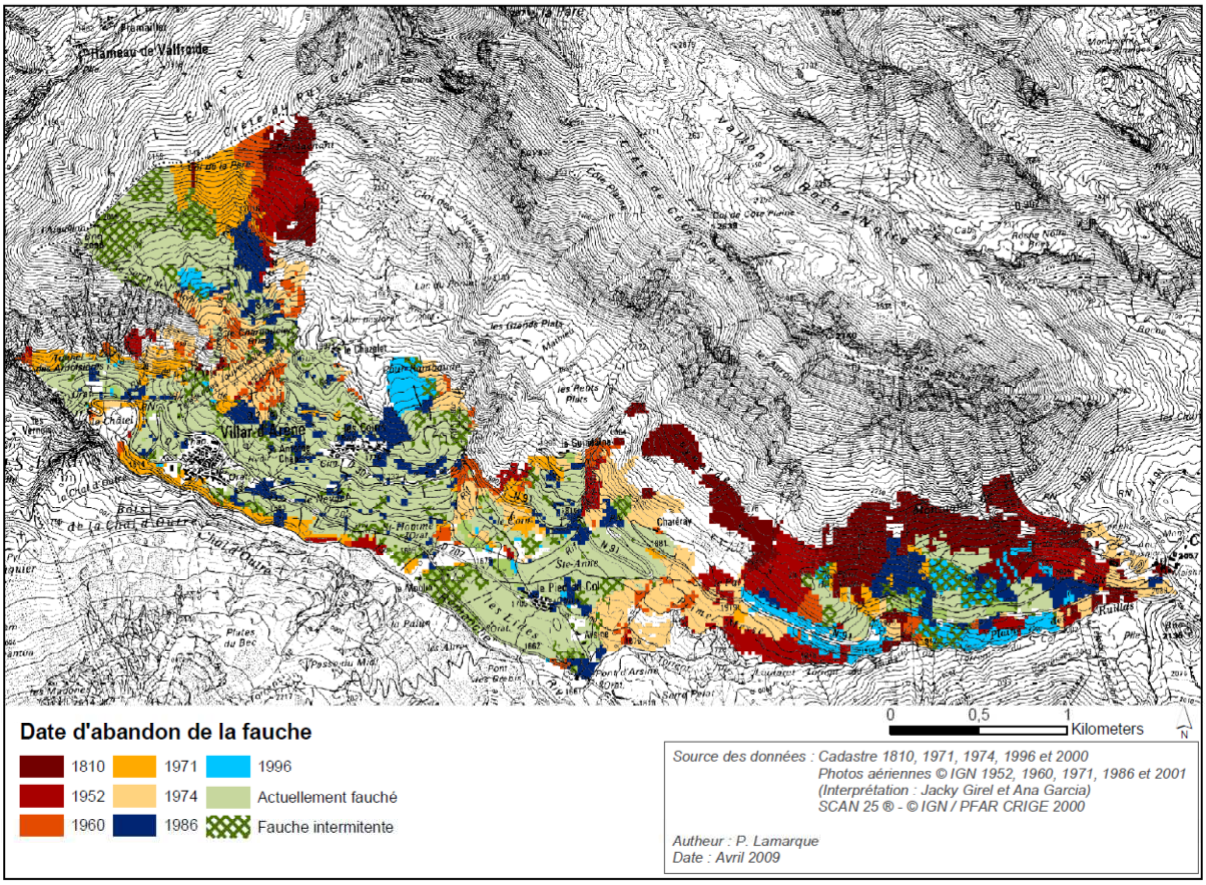


Figure 12 : Dates d'abandon de la fauche identifiée à partir de l'interprétation des photos aériennes de 1952, 1960, 1971, 1974, 1986, 1996 et 2001, et des plans cadastraux de 1810, 1971 et 2001.

Ces changements progressifs de système ont conduit à un important morcellement du foncier qui constituait un handicap important pour les travaux de fenaison et de pâturage. Des échanges officieux ont vite vu le jour entre agriculteurs afin de diminuer le nombre de parcelles enclavées, les distances entre parcelles et les petites surfaces en regroupant plusieurs terrasses en un seul quartier. Afin de régulariser la situation et remplacer les accords verbaux par des baux écrit, l'Association Foncière Pastorale de Villar d'Arène (AFP) a été créée en 1976 et fut la première du département. Cette association a été créée dans un souci d'équitabilité des rentes perçues par les propriétaires des terrains, le prix étant fixé par l'AFP. Dans un premier temps, elle regroupait tous les terrains des parties hautes de l'adret, notamment les terrains communaux pour organiser le pâturage estival entre les agriculteurs. A partir de 1990, la logique de gestion collective a été étendue à l'ensemble de l'espace agricole de l'adret, englobant également les prairies de fauche et les pâturages d'intersaison utilisés au printemps et à l'automne. Les enjeux de l'AFP dépassent la distribution des parcelles entre agriculteurs. Par la régularisation des baux, elle a facilité la contractualisation des aides de la Politique Agricole Commune. De plus des règles sont discutées et fixées en bureau, telles que la date de sortie des animaux sur les parcelles, afin de veiller à la bonne gestion des terrains. Des achats communs de matériel peuvent également être effectués, comme des clôtures ou l'achat de pièges ou de semences de sur-semis suite à l'invasion de campagnols en 2009. Le fonctionnement de l'AFP est supervisé par son bureau qui se compose d'un président, un vice président, un secrétaire et son adjoint, un trésorier, cinq représentants de la commune, deux représentants des propriétaires et deux représentants des agriculteurs. Les propriétaires possèdent un nombre de voix proportionnellement à la surface qu'ils possèdent (une voie = 1ha). La commune est donc majoritaire. Chaque agriculteur, y compris les propriétaires exploitant louent les terrains de fauche 25€/ha et les pâturages d'intersaison 9€/ha. Les alpages sont gérés collectivement par l'intermédiaire du groupement pastoral du Pontet, le prix est de 8€/tête bovine, 2,5€/ tête ovine pour les transhumants et 2€/tête ovine pour les locaux. Les propriétaires touchent quant à eux 23€/ha pour les prés de fauche et 8€/ha pour les pâturages d'intersaison. L'alpage du Pontet est loué 3994€ pour la saison.

Malgré la présence de 9 exploitations, seulement (23 sur l'ensemble du canton), et d'un berger transhumant, l'activité agricole est encore fort présente sur l'adret de Villar d'Arène et ne laisse que très peu de zones abandonnées (ni cultivées, ni fauchées, ni pâturées). Les terrains sont exploités soit en prés de fauche (26%), soit en pâturage d'intersaison (15%), ou en prairies d'alpage (59%), avec quelques terrains labourés résiduels pour la culture familiale de pommes de terre.

Les différents changements d'utilisation des parcelles au cours de l'historique peuvent être traduits en trajectoires d'utilisation du sol (Quétier et al., 2007b). Les catégories de trajectoires des parcelles agricoles (Tableau 1 et figure 13), sont basées sur les changements de gestion entre 1810 - qui est la plus ancienne carte exhaustive du territoire dont on dispose - et la gestion des prairies en 2003 (Girel et al., 2010; Quétier, 2006).

Code des trajectoires	Utilisation passée	Utilisation actuelle	Surface totale (ha)	Altitude (m) (min-max)	Rendement potentiel (T/ha)
11	Terrasse labourée	Terrasse labourée			-
1	Terrasse labourée	Prairie fauchée et fertilisée	71,36	1584 - 1944	5,01
2	Terrasse labourée	Prairie fauchée	117,24	1554 - 1938	3,55
3	Terrasse labourée	Prairie pâturée	109,64	1539 - 1794	2,97
4	Prairie d'altitude fauchée	Prairie d'altitude fauchée	67,72	1854 - 2013	4,24
5	Prairie d'altitude fauchée	Prairie d'altitude pâturée	215,68	1702 - 2024	4,96
7	Alpage	Alpage	381,4	2710 - 2228	2,97

Tableau 1 : Classification des prairies en trajectoires d'utilisation du sol basées sur le passé agricole et l'utilisation actuelle de la parcelle. Surface totale sur le site, altitude et rendements potentiels mesurés selon la biomasse verte récoltée.

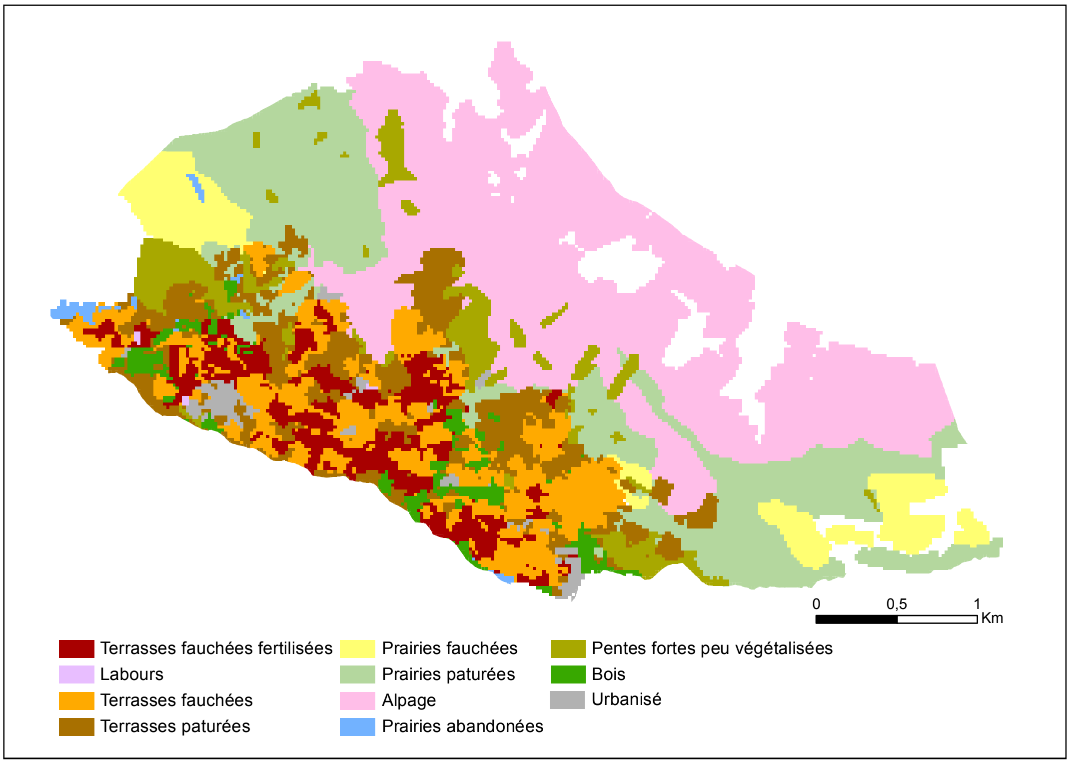


Figure 13 : Répartition spatiale des trajectoires d'utilisation du sol.

4.3.3 Les systèmes d'élevage présents dans les années 2000[3]

L'agriculture actuelle repose essentiellement sur deux types de filières (Deboeuf, 2009). La première concerne l'élevage de génisses de race Abondance ou Tarine à destination des fermes savoyardes et haut-savoyardes des zones de production laitière des appellations d'origine contrôlées (AOC) Beaufort et Reblochon. Les agriculteurs de Savoie et de Haute-Savoie vendent leurs veaux entre 8 et 15 jours afin de préserver leurs ressources en herbe pour les vaches laitières et leur temps de travail, l'élevage des veaux étant contraignant surtout la première année. Ceux-ci sont élevés par les éleveurs faranchins pendant 3 ans avant d'être revendus comme génisses pleines, généralement aux éleveurs d'origine. Autrefois ces transactions s'effectuaient principalement lors des foires d'automne du Chazelet (La Grave) ou de Monêtier-les-Bains alors qu'aujourd'hui la vente et le prix sont généralement fixés bien à l'avance. Toutefois, la rentabilité de ce système tend à baisser car le prix de la génisse n'augmente pas conjointement avec les coûts de production et d'alimentation. Cette tendance a d'ailleurs poussé deux éleveurs faranchins à se convertir progressivement vers un système vaches allaitantes avec développement d'activités de vente directe de viande (veaux ou génisses). Selon eux, ce système valorise beaucoup mieux la vente de bétail que la filière d'élevage de génisses. La seconde filière agricole est l'élevage ovin de viande orienté vers la production d'agneaux vendus en majorité aux coopératives de Gap ou de Briançon sous le label « Agneau de Sisteron » (Indication géographique protégée). Deux agriculteurs interviennent sur les deux filières à la fois, élevant à la fois des génisses et des brebis. Parallèlement à ces deux filières principales, un alpagiste, installé depuis 2000 sur la commune, prend des bêtes en estive et utilise leur lait pour la fabrication de fromages de types gruyère et tomme de Savoie qu'il écoule en vente directe.

Au total à Villar d'Arène, neuf exploitations sont encore en activité actuellement (2009-1012) (Tableau 2). Une des exploitations a son siège sur la commune voisine de La Grave mais exploite certains prés de fauche à Villar d'Arène. Deux agriculteurs sont à la retraite mais conservent leur exploitation, tandis que deux autres sont proches de la retraite (57 et 60 ans). La précarité des bâtiments est un facteur limitant de la reprise pour trois d'entre-elles, ce qui contribuera très probablement à la baisse du nombre d'agriculteurs dans les prochaines années. Les terrains pourront dès lors être redistribués entre agriculteurs restants ou loués à un transhumant. La plupart des agriculteurs, excepté le plus jeune (33 ans) installé depuis 2007, ont repris l'exploitation familiale en changeant parfois légèrement leur orientation d'exploitation vers un agrandissement ou la réduction d'un atelier (ovin ou bovin).

[3] Une description complète pourra être lue dans le rapport d'un stage que j'ai encadré durant cette thèse. Deboeuf, E., 2009, Adaptabilité des systèmes d'élevage de haute-montagne à des aléas. Le cas de Villar d'Arène, Enita de Clermont-Ferrand, France, pp. 91.

Les bâtiments d'exploitation vétustes et peu fonctionnels dans l'ensemble (anciennes bergeries ou étables familiales) sont progressivement modernisés, surtout par les agriculteurs en pleine activité (ni jeune – ni proche de la retraite). Environ la moitié des exploitations sont équipées d'un matériel assez récent, fonctionnel et adapté à la taille des exploitations tandis qu'il est très ancien ou quasi inexistant pour l'autre. Il existe cependant deux Coopératives d'Utilisation du Matériel Agricole (CUMA) sur le canton entre agriculteurs de La Grave et de Villar d'Arène mais elles ne mettent en commun qu'une petite partie du parc matériel des exploitations (herse rotative, tonne à lisier, épandeur à fumier…).

Eleveurs et alpages	Type d'exploitation	Effectif
E1	Elevage de brebis Préalpes communes (agneaux) et génisses d'élevage (abondances et tarines)	170 brebis, 36 génisses et 4 vaches
E2&3	Elevage de brebis Préalpes communes (agneaux) et génisses d'élevage (abondances et tarines)	130 brebis, 10 génisses et 10 vaches
E4&5	Alpagiste : location de vaches laitières (abondances, tarines et montbéliardes) pour la production de fromages (tomme, raclette et gruyère)	15 vaches laitières
E6	Elevage de génisses pour la Savoie	20 vaches, 50 génisses
E7	Elevage de génisses pour la Savoie	75 génisses et 6 vaches
E8	Elevage de brebis Préalpes communes (agneaux)	110 brebis
E9	Elevage de brebis Préalpes communes (agneaux)	50 brebis
A10	Elevage de brebis Préalpes communes (agneaux)	60 brebis
E11	Elevage de génisses pour la Savoie (Eleveur sur La Grave)	135 génisses et 20 vaches
Alpage du Laurichard	Berger : E10	1000 ovins dont 930 transhumants
Alpage de l'Alpe	E6 + d'autres troupeaux de bovins (parcs) : géré par E6 + un troupeau ovin : gardé par 1 berger	270 bovins dont 220 transhumants et 500 ovins
Alpage du Pontet	Parcs	Bovins de la commune
Alpage de Pontet-Chaillol	Eleveur-berger du Vars : EB1	1500 ovins dont 900 transhumants

Tableau 2: Les exploitations et alpages de Villar d'Arène et leurs principales caractéristiques (Deboeuf, 2009).

La majorité des agriculteurs travaillent seuls sur l'exploitation et reçoivent ponctuellement de l'aide de leur entourage proche. Deux éleveurs sont pluriactifs et travaillent dans les travaux publics l'été et le service des voiries (déneigement) ou en station de ski l'hiver. Au niveau des ménages, la pluriactivité est presque la règle puisque le conjoint travaille généralement en dehors de l'exploitation. Ces exploitations peu compétitives par rapport aux exploitations de moyenne montagne et de plaine, sont très dépendantes des aides agricoles notamment de la Politique Agricole Commune (PAC) (voir section 4.3.6). En effet, en haute montagne les charges opérationnelles sont plus importantes car il faut acheter du foin, mais aussi des concentrés et de la paille car il est difficile ou couteux de produire des céréales sur place et la durée d'hivernage est beaucoup plus longue. Le montant des aides équilibre généralement le montant des charges (Figure 14). Par ailleurs, la majorité des exploitations a des annuités d'emprunt sur les bâtiments et le matériel agricole. Dans un contexte d'évolution de ces aides agricoles (révision de la PAC) et d'incertitudes sur les montants de subventions ces exploitations sont donc très fragiles économiquement.

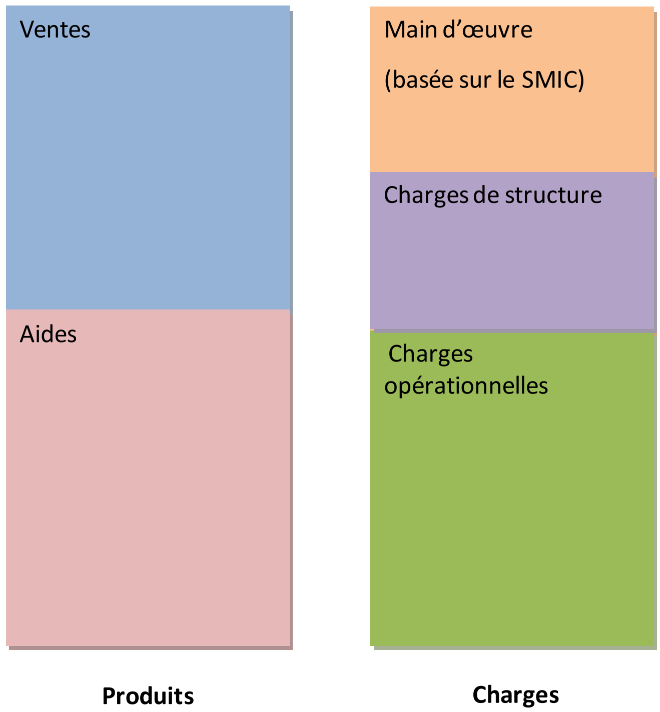


Figure 14: Représentation proportionnelle des résultats économique d'une exploitation de haute montagne. Equilibre entre produits et charges. Basé sur une enquête réalisée auprès de techniciens agricoles.

La fauche occupe une bonne partie du temps de travail estival des éleveurs pendant que le bétail est en alpage. Sur la commune, celle-ci commence début juillet et pour finir entre fin août et fin septembre en fonction des éleveurs (Figure 15). Une seule coupe est réalisée pendant la saison. Les terrains les plus pentus sont encore fauchés à la motofaucheuse. L'altitude écourtant la période végétative et le climat sec empêchent en effet très souvent la fauche du regain. Ce-dernier ne se produisant que sur les parcelles les plus basses, est généralement pâturé de manière très extensive à l'automne pendant un mois après la descente d'alpage. Jusqu'au début des années 2000, l'ensemble des exploitations étaient autonomes pour la production de foin. En revanche, depuis les épisodes de sécheresses (2003-2004 et 2009) et la pullulation de campagnols en 2009, les éleveurs sont contraints de compenser la baisse de production de foin par des achats extérieurs ou par la fauche de parcelles sur d'autres communes. Cette dernière alternative est cependant difficile à mettre en œuvre car des parcelles agricoles mécanisables disponibles sont relativement rares. De plus, cette dernière option n'est pas suffisante pour compléter l'alimentation du bétail pour la période d'hivernage de plus de six mois. Le pâturage n'est quant à lui pas contraint en ressources en raison de la grande surface disponible en alpage ; il ne conduit pas les éleveurs à surpâturer les prairies. Le chargement est compris entre 0,6 et 1 UGB/ha (UGB : Unité Gros Bétail[4]) ou entre 0,02 et 2,1 Jour UGB/ha[5]. L'alpage de Pontet Chaillol est géré par un groupement pastoral constitué des éleveurs ovins. Ce groupement emploie chaque année un berger salarié (entrepreneur de garde) qui vient avec ses 900 brebis (environ) et assure la garde des brebis des acteurs locaux.

En plus des contraintes imposées par le climat et la période de végétation, le rythme agricole de Villar d'Arène est aussi régi par l'Association Foncière Pastorale (AFP). Les dates de mises à l'herbe, fixées par l'AFP, sont identiques d'une année sur l'autre avec une semaine de marge en fonction de la météo : le 8 mai sur les prés d'intersaison pour les ovins, le 20 mai pour les bovins. Sur les alpages communaux la date de montée est le 15 juin et celle de descente le 15 octobre.

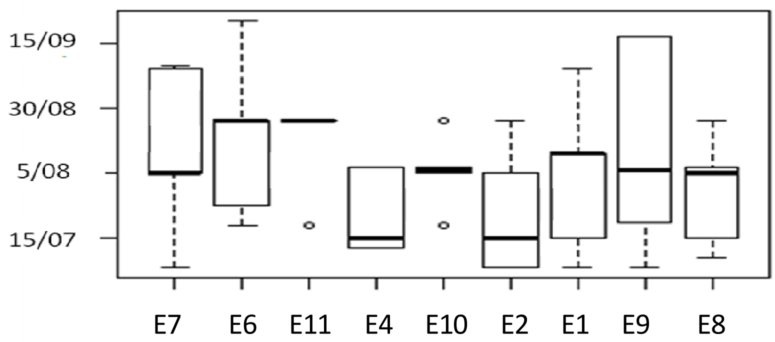


Figure 15 : Etalement de la période de fauche par éleveur allant de début juillet à fin septembre.

La distance entre les terrains exploités par les agriculteurs sur la commune et le siège d'exploitation est au maximum de 8 kilomètres (au niveau du Col du Lautaret). Les prés de fauche s'étendent approximativement entre 1550 et 2000 m d'altitude, les pâturages d'intersaison et les alpages entre 1700 à 2300 m. Le manque de terrains mécanisables (fortes pentes sur la commune) conduisent certains agriculteurs à exploiter des terres (fauche) jusqu'à Bourg d'Oisans (30 km - 700m d'altitude), Auris en Oisans (32 km - 1600 m) ou le Freney d'Oisans (20 km - 1000m). Les agriculteurs exploitent en moyenne 58 ha hors alpage (de 14 à 124 ha).

[4] Calculé selon les normes CORPEN : vache laitières = 1 ; Génisses 0-1 ans = 0,3 ; Génisses 1-2 ans = 0,8, brebis = 0,14.

[5] Jour UGB/ha = ((nombre d'animaux × nombre de jours de pâturage dans l'année) / 365) / surface de la parcelle en hectares)

Le système fourrager des exploitations (en dehors des alpages) est composé majoritairement de prairies de fauche (Figures 16), le reste étant pâturé. Certains prés de fauche reçoivent à l'automne tous les deux à quatre ans le fumier bovin produit sur l'exploitation. L'entièreté du fumier ovin est, quant à lui, récupéré par une entreprise spécialisée (« OvinAlp »). Un agriculteur est équipé d'une fosse à purin qu'il épand chaque année au printemps. Les pâturages d'intersaison et les alpages ne sont jamais fertilisés autrement que par les déjections animales. Le fumier épandu est très concentré (5kg de nitrite/Tonne de fumier frais et 15 unité de potassium / Tonne) car c'est un fumier mou peu pailleux. Celui-ci est épandu à des doses de 10 à 20 tonnes/ha (selon l'estimation d'un technicien agricole) sur les prés ayant une pente inférieure à 18°.

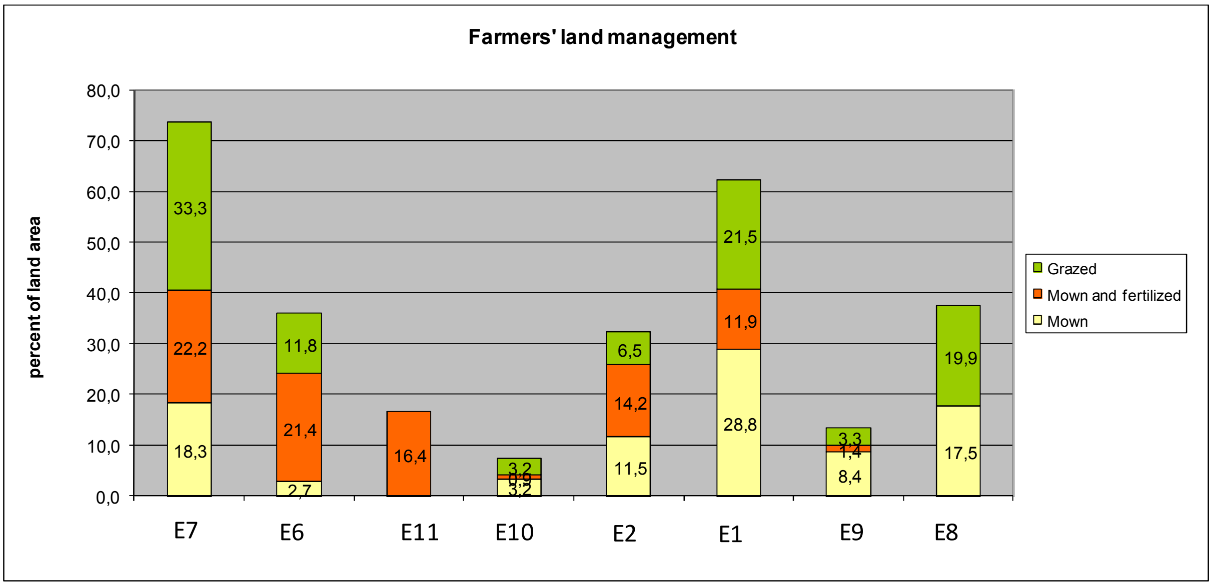


Figure 16: Pourcentage et surface fauchée, fauchée fertilisée et pâturée par agriculteur sur l'adret de Villar d'Arène. E11 a son siège d'exploitation sur la commune voisine et exploite les prairies de Villar d'Arène uniquement comme pâture. E1 et E6 fauchent partiellement sur d'autres communes pour compléter leurs stocks de foin.

4.3.4 La végétation des prairies subalpines

La conjonction des facteurs biophysiques et humains décrits ci-dessus expliquent la présence de communautés végétales différenciées, en termes de composition floristique et de diversité fonctionnelle. Ces différentes communautés végétales correspondent aux trajectoires d'utilisation du sol définies plus haut (Quétier et al., 2007b).

Au sein des terrasses, les prairies fauchées fertilisées sont dominées par des espèces à stratégie exploitatrice qui ont une forte acquisition des ressources, nutriments mais une faible capacité de conservation des nutriments dans leurs tissus comme *Dactylis glomerata*, *Trisetum flavescens*, *Chaerophyllum hirsutum*, *Heracleum sphondylum*, *Geranium sylvaticum*, *Gentiana lutea.* Ce sont ces prairies qui possèdent la plus forte diversité spécifique et fonctionnelle (Quétier et al., 2007b) sur notre site d'étude. Malgré la fertilisation, le niveau de disponibilité minérale de ces prairies reste en effet relativement bas comparativement à celui de prairies fertilisées de plaine, ce qui explique cette particularité des prairies fertilisées de notre site d'étude (Quétier et al., 2007b). A l'inverse, lorsque ces prairies n'ont pas été fertilisées, elles sont dominées par des espèces à stratégie conservatrice qui ont une forte capacité à conserver les nutriments dans leurs tissus comme *Briza media*, *Bromus erectus*, *Sesleria caerulea, Festuca ovina.* Ces prairies sont légèrement moins riches en terme de diversité spécifique et fonctionnelle que les prairies fertilisées.

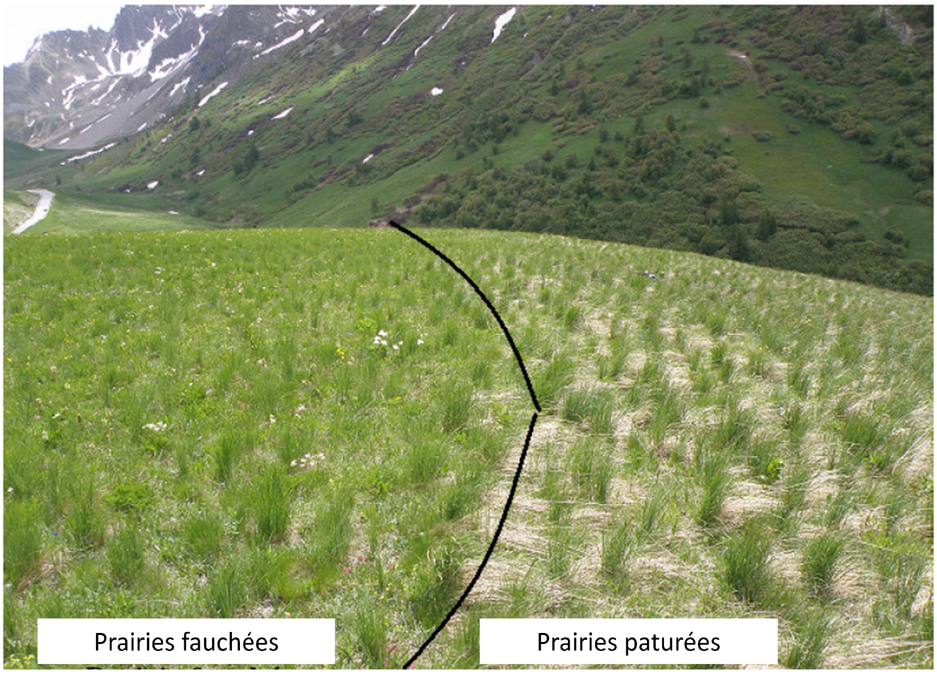


Figure 17 : Influence de la fauche sur les prairies à Fétuque paniculée (Queyrellins) Photo S.Caubet, mai 2009

Les prés de fauche n'ayant jamais été labourés sont dominés par *Festuca paniculata* (Figure 17). Lorsque la fauche est maintenue, les prairies sont dominées par des espèces à stratégie conservatrice comme *Festuca paniculata*, *Meum athamanticum*, *Trifolium alpinum*, *Festuca rubra*, *Sanguisorba officinalis* et certaines espèces exploitatrices. Enfin, quand la pâture estivale prend la place de la fauche sur les parcelles les moins accessibles et les plus difficilement mécanisables, *Festuca paniculata* (appelée Quéyrelle dans la région) devient fortement dominante car elle est très peu appétente (> 70 % de la biomasse de la communauté) (Quétier et al., 2012; Quétier et al., 2007b). Cette espèce très compétitrice (Gross et al., 2010a) produit beaucoup de litière peu dégradable au niveau des sols (Vittoz et al., 2005), ce qui ralentit les cycles des nutriments lorsqu'elle domine.

4.3.5 Le tourisme[6]

Le col du Lautaret est un axe important de communication entre Alpes du Nord et Alpes du Sud. Le milieu naturel exceptionnel qui entoure le Col et la vue sur la Meije et ses glaciers fait de ce site un lieu touristique des Alpes très fréquenté, été comme hiver. Le tourisme est l'activité économique principale du canton malgré que celui-ci soit saisonnier, avec l'hiver comme saison principale sur le plan des retombées économiques. Les activités de loisirs y sont multiples en hiver comme en été. En hiver le ski *freeride*, le ski de randonnée, le *snowkite* et les balades en raquettes sont les principales activités. L'été, l'alpinisme et la randonnée constituent les principales activités sportives, mais de nombreux touristes viennent en voiture ou vélo pour contempler la vue et passer au col du Lautaret avant de rejoindre le mythique col du Galibier.

L'été, le tourisme de Villar est plutôt un tourisme de passage constitué pour moitié de familles et pour autre moitié de couples (selon enquêtes). La moitié des touristes ne reste qu'une journée sur place (50% des enquêtés) malgré les 1453 lits touristiques disponibles sur la commune de La Grave et 451 pour la commune de Villar d'Arène (Vivant, 2007). Un tiers des touristes résident durant leurs vacances dans la vallée de la Guisane (Serre-chevalier), un autres tiers a parcouru entre 30 et 73 km (provenance Valloire, Valmenier, les Deux-Alpes, Bourg D'Oisans, l'Alpe d'Huez) pour venir visiter ou randonner sur la commune. Ceux-ci viennent principalement pour le cadre naturel (80% des enquêtés) estimant que la beauté du site est majoritairement due à la présence de glaciers et du relief montagneux (50 % des enquêtés), la nature sauvage et le paysage agricole, la présence de fleurs ou simplement la couleur verte des prés contrastant avec les glaciers constituent des éléments secondaires mais importants. Des craintes sont cependant émises par les touristes sur la perte du caractère naturel dû à l'urbanisation comme dans d'autres vallées voisines. Le canton de La Grave qui n'a pas cédé à la tentation des grands chantiers touristiques comme les communes de Serre-Chevalier de l'autre coté du col du Lautaret ou de l'Alpe d'Huez et les 2 Alpes du coté Isère bénéficient maintenant d'un atout paysager essentiel.

En ce qui concerne les potentialités agri-touristiques, les touristes enquêtés ont montré un intérêt pour l'achat de produit régionaux (65%) (« à condition de ne pas se faire arnaquer »), et dans une moindre mesure pour des visites d'exploitations (20%) ou des manifestations agricoles faisant découvrir l'agriculture traditionnelle et les produits locaux (30%).

[6] La plupart des données proviennent d'une enquête que j'ai réalisée dans le cadre de cette thèse durant l'été 2011 auprès de 90 touristes sur la commune de Villar d'Arène.

4.3.6 Soutien à l'agriculture et à la conservation de la biodiversité

La politique agricole commune

La politique agricole commune (PAC) a été mise en place en 1962 dans le but de garantir l'autosuffisance alimentaire de l'Europe. La réussite de cet objectif atteint dès les années 1980, s'est accompagnée d'effets pervers (ex. excédent de production, explosion du budget agricole européen). Ceci a conduit les institutions européennes à réformer la PAC à plusieurs reprises de façon importante : (1) en 1992, réforme de « Mac Sharry » mettant en place les aides directes et les mesures agri-environnementales; (2) en 1999, l' « Agenda 2000 » instaurant un deuxième pilier (Figure 18) dédié au développement rural qui se concrétise en France par la mise en place des contrats territorial d'exploitation (CTE) ; (3) en 2003, les « accords de Luxembourg » conduisant au découplage et à la conditionnalité des aides. Le découplage des aides consiste à rendre le montant des aides indépendants des quantités produites (surface cultivées ou quantité d'animaux détenus). Cependant, la France ayant fait le choix de re-coupler certaines aides, une part des aides versées à l'agriculteur reste proportionnelle à la production, alors que l'autre part est indépendante de la production et est appelée droit à payement unique (DPU). Le montant du DPU est fixé en fonction de la surface moyenne déclarée et de la moyenne des aides perçues sur la période 2000-2002. Le dispositif de conditionnalité des aides soumet le versement de certaines aides communautaires au respect d'exigences de base en matière d'environnement, de bonnes conditions agricoles et environnementales (BCAE), de santé (santé publique, santé des animaux, santé des végétaux) et de protection animale.

En vue de la prochaine réforme en 2013, un bilan de santé de la PAC a été réalisé en 2009 et a introduit de nouvelles mesures telles que la suppression progressive du couplage des aides et des quotas laitiers. La commission européenne a déjà esquissé trois scénarios de réforme pour 2013, combinant des paiements directs, des mesures de marché et des mesures visant le développement rural (Commission Européenne, 2010). Quel que soit le scénario, les paiements directs seraient justifiés (totalement ou en partie) par la production par les agriculteurs de « biens publics environnementaux » : paysages visuels, biodiversité liée aux milieux ouverts etc. Les instruments de type mesure agri-environnementale (MAE) sont donc appelés à se généraliser (Deverre and de Sainte Marie, 2008). Par conséquent, il y a lieu d'anticiper la nécessité croissante d'une caractérisation fine de ces « biens publics environnementaux » et des pratiques agricoles qui sont compatibles avec leur production (c'est le domaine de l'éco-conditionnalité) ou qui la modifient, à la hausse ou à la baisse.

Actuellement, deux mécanismes de financement sous-tendent la PAC. Le premier pilier concerne le soutien des marchés et des prix agricoles. Le deuxième pilier consacré au développement rural se concentre sur la multifonctionnalité de l'agriculture, la promotion de la protection de l'environnement dans les milieux agricoles et la participation au développement durable.

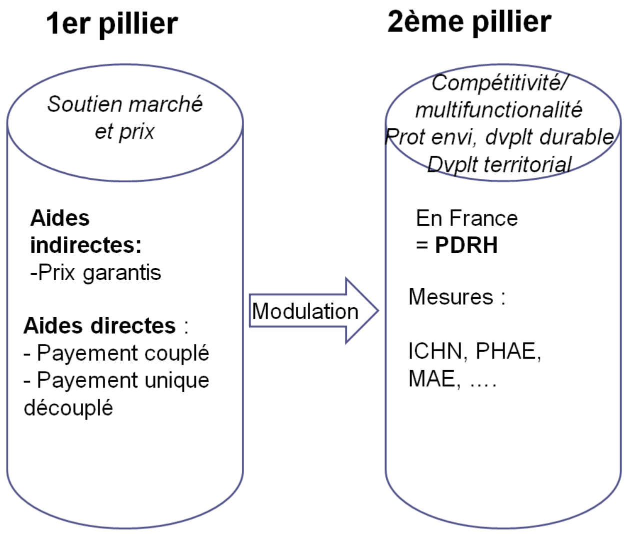


Figure 18: Les deux piliers de la PAC.

Toutes ces aides ont pour objectifs de : (i) fournir un complément de revenus aux exploitants agricoles, (ii) de maintenir une activité agricole dans les zones défavorisées, (iii) de contribuer à l'entretien de l'environnement. Toutefois, à l'heure actuelle peu de services écosystémiques sont pris en compte explicitement dans ces mesures (Grard, 2010).

A Villar d'Arène, les principales aides perçues par les agriculteurs sont :

- Les Indemnités compensatoires de Handicaps Naturels (ICHN) qui contribuent au maintien d'une activité agricole viable dans les zones fragiles et à la préservation des écosystèmes diversifiés et des caractéristiques paysagères de l'espace agraire de ces zones tels que les zones de hautes montagnes dont fait partie Villar d'Arène. Cette aide est indispensable pour compenser les difficultés structurelles auxquelles sont confrontées les exploitations agricoles situées en zone défavorisée et ainsi y maintenir une activité économique souvent essentielle. La surface primée est plafonnée à 50 hectares pour un agriculteur. Cette aide impose un chargement minimal de 0,3 UGB/ha, sinon une diminution de 10% des aides est effectuée.
- La prime Herbagère AgroEnvironnementale (PHAE-2) vise à maintenir les surfaces en herbe et d'entretenir les espaces à gestion extensive et à maintenir les activités de fauche ou de pâture. Leur montant unitaire est de 76€/ha/an. Elle impose un chargement compris entre 0,01 UGB/ha et 1,4 UGB/ha.

- La prime au maintien des troupeaux de vaches allaitantes (PMTVA) a pour objectif de maintenir la production de veaux nourris par leurs mères, en soutenant économiquement les producteurs de bovins de race à viande
- La prime à la brebis et à la chèvre (PBC) fait partie des régimes de soutien direct aux filières ovine et caprine structurellement fragiles. Le montant de base de l'aide aux ovins est de 20,76 € / tête .

Si ces aides constituent un atout pour les exploitations en apportant une part importante du revenu des éleveurs, elles peuvent aussi être source de contraintes par leurs conditions d'obtention telles que les conditions concernant le chargement, les « droits », l'environnement.

- Les conditions de chargement ont pour but d'inciter le développement de systèmes de production plus extensifs en diminuant la densité animale sur l'exploitation en limitant le nombre d'UGB/ha. Toutefois, certaines aides imposent aussi un seuil minimal qui peut-être difficile à tenir dans le cadre d'exploitation très extensives comme à Villar d'Arène.
- Pour la PMTVA et la PBC, chaque producteur dispose de « droits », mais la gestion des droits est effectuée à l'échelle départementale qui dispose d'un nombre maximum de droits
- Les conditions environnementales concernent par exemple la protection des eaux contre la pollution par les nitrates en définissant les quantités de fumier épandue (limité à 170 kg de nitrate/ha), les périodes d'épandages, les distances au point d'eau et les conditions de stockage au travers d'un plan prévisionnel de fumure et d'un cahier d'enregistrement des pratiques d'épandage.

Natura 2000 et le Parc National des écrins

Les sites Natura 2000 sont définis par les états membres de l'Union européenne sur la base de la présence d'habitats naturels et semi-naturels considérés comme importants pour la conservation de la biodiversité à l'échelle européenne avec pour objectif de les maintenir dans un état de conservation favorable (Article 1er Directive 92/43/CEE). Le site Natura 2000 « FR9301498 - COMBEYNOT - LAUTARET – ECRINS », qui comprend quasi la totalité de la zone d'étude, a été désigné pour les nombreux habitats alpins et subalpins figurant parmi la liste de ces habitats importants (habitats dits d' « intérêt communautaire » figurant dans l'Annexe I de la Directive) en s'appuyant sur les limites existantes du Parc national des Ecrins. Parmi ces habitats importants, figurent les prairies de fauche de montagne dont la plupart relève de l'habitat Natura 2000 « 6520 - Prairies de fauche de montagne ». Ce site Natura 2000 possède par conséquent une responsabilité pour la conservation de ces prairies de fauche de montagne en

Europe, plus particulièrement pour les prairies de fauche d'altitude, atteignant 2.000 m sur le site, devenues assez rares à l'échelle des Alpes françaises. Cette responsabilité implique de maintenir ces prairies dans un état de conservation qui correspond à des communautés mésotrophes fauchées extensivement riches en espèces, comprenant notamment de nombreuses dicotylédones à floraison abondante : *Geranium sylvaticum, Polygonum bistorta, Campanula rhomboidalis, Phyteuma spicatum, Astrantia major, Trollius europaeus, Pimpinella major, Meum athamanticum, Crepis pyrenaica* (European Commission, 2007).

Sur le terrain, la mise en œuvre de ces mesures de conservation est assurée par le Parc National des Ecrins qui est chargé de définir en amont, en collaboration avec les services de l'état, un cadre des mesures de gestion à proposer aux agriculteurs (appelé « Mesure agri-environnementale territorialisée » - dite MAET). Dans le cadre de cette MAET portée par le parc, les agriculteurs ont la possibilité de contractualiser un « contrat Natura 2000 » pour le report de la date de fauche ainsi que l'application d'un cahier des charges particulier sur l'entretien, en échange d'une rémunération. Parallèlement, se développent dans d'autres régions des mesures agro-environnementales visant une obligation de résultats, plus que de moyens, envers la gestion des prairies fauchées. Pour être éligible, une prairie doit contenir un minimum de plantes d'une liste de plantes indicatrices des régimes extensifs de fauche. Cette mesure dite « prairies-fleuries » n'est pas encore mise en place dans la zone d'étude mais un concours organisé en 2010 a déjà sensibilisé la population agricole.

Références

Aarnink, W., Bunning, S., Collette, L., Mulvany, P., 1999, Sustaining agricultural biodiversity and agro-ecosystem functions: Opportunities, incentives and approaches for the conservation and sustainable use of agricultural biodiversity in agro-ecosystems and production systems. An International Technical Workshop FAO, Rome (Italy). Sustainable Development Dept., Rome.

Anderson, B. J., Armsworth, P. R., Eigenbrod, F., Thomas, C. D., Gillings, S., Heinemeyer, A., Roy, D. B., Gaston, K. J., 2009, Spatial covariance between biodiversity and other ecosystem service priorities, *Journal of Applied Ecology* **46**(4)**:**888-896.

Bakker, M. M., Govers, G., van Doorn, A., Quetier, F., Chouvardas, D., Rounsevell, M., 2008, The response of soil erosion and sediment export to land-use change in four areas of Europe: The importance of landscape pattern, *Geomorphology* **98:**213-226.

Barrera-Bassols, N., Zinck, J. A., 2003, Ethnopedology: a worldwide view on the soil knowledge of local people, *Geoderma* **111**(3-4)**:**171-195.

Beniston, M., 2000, Environmental change in mountains and uplands, Arnold, London.

Berkes, F., Colding, J., Folke, C., 2000, Rediscovery of traditional ecological knowledge as adaptive management, *Ecological Applications* **10**(5)**:**1251-1262.

Bingham, G., Bishop, R., Brody, M., Bromley, D., Clark, E., Cooper, W., Costanza, R., Hale, T., Hayden, G., Kellert, S., Norgaard, R., Norton, B., Payne, J., Russell, C., Suter, G., 1995, ISSUES IN ECOSYSTEM VALUATION - IMPROVING INFORMATION FOR DECISION-MAKING, *Ecological Economics* **14**(2)**:**73-90.

Boyd, J., Banzhaf, S., 2007, What are ecosystem services? The need for standardized environmental accounting units, *Ecological Economics* **63**(2-3)**:**616-626.

Buijs, A. E., Fischer, A., Rink, D., Young, J. C., 2008, Looking beyond superficial knowledge gaps: Understanding public representations of biodiversity, *International Journal of Biodiversity Science & Management* **4**(2)**:**65 - 80.

Carpenter, S. R., Bennett, E. M., Peterson, G. D., 2006, Scenarios for ecosystem services: An overview, *Ecology and society* **11**(1)**:**14.

Carpenter, S. R., Mooney, H. A., Agard, J., Capistrano, D., DeFries, R. S., Diaz, S., Dietz, T., Duraiappah, A. K., Oteng-Yeboah, A., Pereira, H. M., Perrings, C., Reid, W. V., Sarukhan, J., Scholes, R. J., Whyte, A., 2009, Science for managing ecosystem services: Beyond the Millenium Ecosytem Assessment, *Proceedings of the National Academy of Science* **106**(5)**:**1305-1312.

Chapin, F. S., Zavaleta, E. S., Eviner, V. T., Naylor, R. L., Vitousek, P. M., Reynolds, H. L., Hooper, D. U., Lavorel, S., Sala, O. E., Hobbie, S. E., Mack, M. C., Diaz, S., 2000, Consequences of changing biodiversity, *Nature* **405**(6783)**:**234-242.

Chevalier, J., Buckles, D., 2009, SAS²: Guide sur la recherche collaborative et l'engagement social (ESKA, ed.), Ottawa, Canada, pp. 363.

Cheveau, M., Imbeau, L., Drapeau, P., Belanger, L., 2008, Current status and future directions of traditional ecological knowledge in forest management: a review, *Forestry Chronicle* **84**(2)**:**231-243.

CIPRA Internationale Alpenschutzkommission, 2001, 2ème Rapport sur l'état des Alpes. Données, faits, problèmes, esquisses de solutions, CIPRA Internationale, Schaan (Liechtenstein), pp. 424.

Collins, S. L., Carpenter, S. R., Swinton, S. M., Orenstein, D. E., Childers, D. L., Gragson, T. L., Grimm, N. B., Grove, M., Harlan, S. L., Kaye, J. P., Knapp, A. K., Kofinas, G. P., Magnuson, J. J., McDowell, W. H., Melack, J. M., Ogden, L. A., Robertson, G. P., Smith, M. D., Whitmer, A. C., 2011, An integrated conceptual framework for long-term social-ecological research, *Frontiers in Ecology and the Environment* **9**(6)**:**351-357.

Commission Européenne, 2010, Communication de la commission au parlement européen, au conseil, au comité économique et social européen et au comité des régions - La PAC à l'horizon 2010 : Alimentation, ressources naturelles et territoires - relever les défis de l'avenir, Commission Européenne, Bruxelles, Belgique.

Cornelissen, J., Lavorel, S., Garnier, E., Diaz, S., Buchmann, N., Gurvich, D., Reich, P. B., Ter Steege, H., Morgan, H., Van Der Heijden, M., 2003, A handbook of protocols for standardised and easy measurement of plant functional traits worldwide, *Australian Journal of Botany* **51**(4)**:**335-380.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R. V., Paruelo, J., Raskin, R. G., Sutton, P., van den Belt, M., 1997, The value of the worl's ecosystem services and natural capital, *Nature* **387:**253-260.

Cowling, R. M., Egoh, B., Knight , A. T., O'Farrell, P. J., Reyers, B., Rouget, M., Roux, D. J., Welz, A., Wilhelm-Rechman, A., 2008, An operational model for mainstreaming ecosystem services for implementation, *Proceedings of the National Academy of Science* **105**(28)**:**9483-9488.

Daily, G. C., 1997, Nature's Services: Societal Dependence on Natural Ecosystems, Island Press, Washington D.C.

de Bello, F., Lavorel, S., Diaz, S., Harrington, R., Cornelissen, J. H. C., Bardgett, R. D., Berg, M. P., Cipriotti, P., Feld, C. K., Hering, D., da Silva, P. M., Potts, S. G., Sandin, L., Sousa, J. P., Storkey, J., Wardle, D. A., Harrison, P. A., 2010, Towards an assessment of multiple ecosystem processes and services via functional traits, *Biodiversity and Conservation* **19**(10)**:**2873-2893.

De Deyn, G. B., Cornelissen, J. H. C., Bardgett, R. D., 2008, Plant functional traits and soil carbon sequestration in contrasting biomes, *Ecology letters* **11**(5)**:**516-531.

de Groot, R. S., Alkemade, R., Braat, L., Hein, L., Willemen, L., 2010, Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making, *Ecological Complexity* **7**(3)**:**260-272.

Deboeuf, E., 2009, Adaptabilité des systèmes d'élevage de haute-montagne à des aléas. Le cas de Villar d'Arène, Enita de Clermont-Ferrand, France, pp. 91.

Deverre, C., de Sainte Marie, C., 2008, L'Ecologisation de la politique agricole européenne. Verdissement ou refondation des systèmes agro-alimentaires?, *Cahiers d'Economie et Sociologie Rurales* **89**(4)**:**83-104.

Diaz, S., Fargione, J., Stuart Chapin, F., Tilman, D., 2006, Biodiversity Loss Threatens Human Well-Being, *PLoS Biology* **4**(8)**:**1300-1305.

Diaz, S., Lavorel, S., de Bello, F., Quétier, F., Grigulis, K., Robson, T. M., 2007, Incorporating plant functional diversity effects in ecosystem service assessments, *Proceedings of the National Academy of Science* **104**(52)**:**20684-20689.

Diaz, S., Quetier, F., Caceres, D. M., Trainor, S. F., Perez-Harguindeguy, N., Bret-Harte, M. S., Finegan, B., Pena-Claros, M., Poorter, L., 2011, Linking functional diversity and social actor strategies in a framework for interdisciplinary analysis of nature's benefits to society, *Proceedings of the National Academy of Sciences of the United States of America* **108**(3)**:**895-902.

Doré, T., Makowski, D., Malézieux, E., Munier-Jolain, N., Tchamitchian, M., Tittonell, P., 2011, Facing up to the paradigm of ecological intensification in agronomy: Revisiting methods, concepts and knowledge, *European Journal of Agronomy* **34**(4)**:**197-210.

Egoh, B., Reyers, B., Rouget, M., Richardson, D. M., Le Maitre, D. C., van Jaarsveld, A. S., 2008, Mapping ecosystem services for planning and management, *Agriculture, Ecosystems and Environment* **127:**135-140.

Ehrlich, P. R., Mooney, H. A., 1983, EXTINCTION, SUBSTITUTION, AND ECOSYSTEM SERVICES, *Bioscience* **33**(4)**:**248-254.

Eigenbrod, F., Armsworth, P. R., Anderson, B. J., Heinemeyer, A., Gillings, S., Roy, D. B., Thomas, C. D., Gaston, K. J., 2010, The impact of proxy-based methods on mapping the distribution of ecosystem services, *Journal of Applied Ecology* **47**(2)**:**377-385.

Etienne, M., (coord), 2010, La modélisation d'accompagnement: Une démarche participative en appui au développement durable, Quae Editions, Versailles, France, pp. 368.

European Commission, 2007, Interpretation manual of european union habitats (D. E. European Commission, Nature and biodiversity, ed.), Bruxelles, pp. 142.

Fischer, A., Young, J. C., 2007, Understanding mental constructs of biodiversity: Implications for biodiversity management and conservation, *Biological Conservation* **136**(2)**:**271-282.

Folke, C., 2006, Resilience: The emergence of a perspective for social-ecological systems analyses, *Global environmental change* **16**(3)**:**253-267.

Gibon, A., 2005, Managing grasland for production, the environment and the landscape. Challenges at the farm and the landscape level, *Livestock production science* **96:**11-31.

Girel, J., Quétier, F., Bignon, A., Aubert, S., 2010, Histoire de l'agriculture en Oisans. Hautes Romanche et pays faranchin. Villar d'Arène, Hautes-Alpes, in: *La Galerie de l'Alpe*, Station Alpine Joseph Fourier, Grenoble, France, pp. 79.

Gómez-Baggethun, E., de Groot, R., Lomas, P. L., Montes, C., 2010, The history of ecosystem services in economic theory and practice: From early notions to markets and payment schemes, *Ecological Economics* **69**(6)**:**1209-1218.

Grard, M., 2010, Le rôle des politiques publiques dans les services écosystémiques des prairies de montagne, Master sciences et politiques de l'environnement, UMPC-IEP, Paris.

Gross, N., Liancourt, P., Choler, P., Suding, K., Lavorel, S., 2010, Strain and vegetation effects on local limiting resources explain the outcomes of biotic interactions, *Perspectives in Plant Ecology, Evolution and Systematics* **12:**9-19.

Haberl, H., Winiwarter, V., Andersson, K., Ayres, R. U., Boone, C., Castillo, A., Cunfer, G., Fischer-Kowalski, M., Freudenburg, W. R., Furman, E., Kaufmann, R., Krausmann, F., Langthaler, E., Lotze-Campen, H., Mirtl, M., Redman, C. L., Reenberg, A., Wardell, A., Warr, B., Zechmeister, H., 2006, From LTER to LTSER: Conceptualizing the socioeconomic dimension of long-term socioecological research, *Ecology and Society* **11**(2).

Haines-Young, R., Potschin, M., 2010, The links between biodiversity, ecosystem services and human well-being, in: *Ecosystem ecology: A New Synthesis* (D. Raffaelli, C. Frid, eds.), CUP, Cambridge.

Harrington, R., Anton, C., Dawson, T. P., de Bello, F., Feld, C. K., Haslett, J. R., Kluvankova-Oravska, T., Kontogianni, A., Lavorel, S., Luck, G. W., Rounsevell, M. D. A., Samways, M. J., Settele, J., Skourtos, M., Spangenberg, J. H., Vandewalle, M., Zobel, M., Harrison, P. A., 2010, Ecosystem services and biodiversity conservation: concepts and a glossary, *Biodiversity and Conservation* **19**(10)**:**2773-2790.

Hein, L., van Koppen, K., de Groot, R. S., van Ierland, E. C., 2006, Spatial scales, stakeholders and the valuation of ecosystem services, *Ecological Economics* **57**(2)**:**209-228.

Hooper, D. U., Chapin, F. S., Ewel, J. J., Hector, A., Inchausti, P., Lavorel, S., Lawton, J. H., Lodge, D. M., Loreau, M., Naeem, S., Schmid, B., Setala, H., Symstad, A. J., Vandermeer, J., Wardle, D. A., 2005, Effects of biodiversity on ecosystem functioning: A consensus of current knowledge, *Ecological Monographs* **75**(1)**:**3-35.

Kazakou, E., Vile, D., Shipley, B., Gallet, C., Garnier, E., 2006, Coâ€variations in litter decomposition, leaf traits and plant growth in species from a Mediterranean oldâ€field succession, *Functional Ecology* **20**(1)**:**21-30.

Kelemen, Gomez-Baggethun, E., in revision, Participatory Methods for Valuing Ecosystem Services, *THEMES Summer School, Lisbon, May 2008*.

Koellner, T., 2008, Supply and demand for ecosystem services in mountainous regions, in: *COST Strategic Conference Global Change and Sustainable Development in Mountain Regions*, Innsbruck university press, Innsbruck, pp. 61-70.

Krutilla, J. V., 1967, Conservation reconsidered, *American Economic Review* **57:**777-786.

Lambin, E. F., Meyfroidt, P., 2010, Land use transitions: Socio-ecological feedback versus socio-economic change, *Land Use Policy* **27**(2)**:**108-118.

Lambin, E. F., Turner, B. L., Geist, H. J., Agbola, S. B., Angelsen, A., Bruce, J. W., Coomes, O. T., Dirzo, R., Fischer, G., Folke, C., George, P. S., Homewood, K., Imbernon, J., Leemans, R., Li, X. B., Moran, E. F., Mortimore, M., Ramakrishnan, P. S., Richards, J. F., Skanes, H., Steffen, W., Stone, G. D., Svedin, U., Veldkamp, T. A., Vogel, C., Xu, J. C., 2001, The causes of land-use and land-cover change: moving beyond the myths, *Global Environmental Change-Human and Policy Dimensions* **11**(4)**:**261-269.

Lautenbach, S., Kugel, C., Lausch, A., Seppelt, R., 2011, Analysis of historic changes in regional ecosystem service provisioning using land use data, *Ecological Indicators* **11**(2)**:**676-687.

Lavorel, S., Garnier, E., 2002, Predicting changes in community composition and ecosystem functionning from plant traits: revisiting the Holy Grail, *Functional Ecology* **16:**545-556.

Lavorel, S., Grigulis, K., 2012, How fundamental plant functional trait relationships scale-up to trade-offs and synergies in ecosystem services, *Journal of Ecology* **100**(1)**:**128-140.

Lavorel, S., Grigulis, K., Lamarque, P., Colace, M.-P., Garden, D., Girel, J., Pellet, G., Douzet, R., 2011, Using plant functional traits to understand the landscape distribution of multiple ecosystem services, *Journal of Ecology* **99**(1)**:**135-147.

Lavorel, S., McIntyre, S., Landsberg, J., Forbes, T. D. A., 1997, Plant functional classifications: from general groups to specific groups based on response to disturbance, *TREE* **12**(12)**:**474-478.

Lavorel, S., Spiegelberger, T., Mauz, I., Bigot, S., Granjou, C., Dobremez, L., Nettier, B., Thuiller, W., Brun, J.-J., Cozic, P., 2012, Coupled long-term dynamics of climate, land use, ecosystems and ecosystem services in the Central French Alps, in: *Long term socio-ecological research: Studies in society-nature interactions across spatial and temporal scales (S. J. Singh, H. Haberl, M. Chertow, M. Mirtl, M. Schmid, eds.)* (Springer-Verlag, ed.), Springer-Verlag.

Lemaire, G., Wilkins, R., Hodgson, J., 2005, Challenges for grassland science: managing research priorities, *Agriculture, ecosystems & environment* **108**(2)**:**99-108.

Lewan, L., Soderqvist, T., 2002, Knowledge and recognition of ecosystem services among the general public in a drainage basin in Scania, Southern Sweden, *Ecological Economics* **42**(3)**:**459-467.

Lindemann-Matthies, P., Junge, X., Matthies, D., 2010, The influence of plant diversity on people's perception and aesthetic appreciation of grassland vegetation, *Biological Conservation* **143**(1)**:**195-202.

Liu, J., Dietz, T., Carpenter, S. R., Folke, C., Alberti, M., Redman, C. L., Schneider, S. H., Ostrom, E., Pell, A. N., Lubchenco, J., Taylor, W. W., Ouyang, Z., Deadman, P., Kratz, T., Provencher, W., 2007a, Coupled Human and Natural Systems, *AMBIO: A Journal of the Human Environment* **36**(8)**:**639-649.

Liu, J. G., Dietz, T., Carpenter, S. R., Alberti, M., Folke, C., Moran, E., Pell, A. N., Deadman, P., Kratz, T., Lubchenco, J., Ostrom, E., Ouyang, Z., Provencher, W., Redman, C. L., Schneider, S. H., Taylor, W. W., 2007b, Complexity of coupled human and natural systems, *Science* **317**(5844)**:**1513-1516.

Liu, J. G., Dietz, T., Carpenter, S. R., Folke, C., Alberti, M., Redman, C. L., Schneider, S. H., Ostrom, E., Pell, A. N., Lubchenco, J., Taylor, W. W., Ouyang, Z. Y., Deadman, P., Kratz, T., Provencher, W., 2007c, Coupled human and natural systems, *Ambio* **36**(8)**:**639-649.

MacDonald, D., Crabtree, J. R., Wiesinger, G., Dax, T., Stamou, N., Fleury, P., Gutierrez Lazpita, J., Gibon, A., 2000, Agricultural abandonment in mountain areas of Europe: Environmental consequences and policy response, *Journal of Environmental Management* **59:**47-69.

Mace, G. M., Bateman, I., (ed.), 2011, Conceptual Framework and Methodology. In: The UK National Ecosystem Assessment Technical Report. UK National Ecosystem Assessment, UNEP-WCMC, Cambridge.

Mace, G. M., Cramer, W., Díaz, S., Faith, D. P., Larigauderie, A., Le Prestre, P., Palmer, M., Perrings, C., Scholes, R. J., Walpole, M., Walther, B. A., Watson, J. E. M., Mooney, H. A., 2010, Biodiversity targets after 2010, *Current Opinion in Environmental Sustainability* **In Press, Corrected Proof**.

Marsden, T., Sonnino, R., 2008, Rural development and the regional state: Denying multifunctional agriculture in the UK, *Journal of Rural Studies* **24**(4)**:**422-431.

Martinez-Harms, M. J., Balvanera, P., 2012, Methods for mapping ecosystem service supply: a review, *International Journal of Biodiversity Science, Ecosystem Services & Management***:**1-9.

Maurer, K., Weyand, A., Fischer, M., Stacklin, J., 2006, Old cultural traditions, in addition to land use and topography, are shaping plant diversity of grasslands in the Alps, *Biological Conservation* **130**(3)**:**438-446.

MEA, 2005, Millennium Ecosystem Assessment. Ecosystems and Human Well-being: Synthesis, Island Press, Washington DC U.S.A.

Menzel, S., Teng, J., 2009, Ecosystem Services as a Stakeholder-Driven Concept for Conservation Science, *Conservation Biology* **24**(3)**:**907-909.

Meyfroidt, P., 2012, Environmental cognitions, land change, and social-ecological feedbacks: an overview, *Journal of Land Use Science*.

Meyfroidt, P., in revision, Environmental cognitions, land change and social-ecological feedbacks: local case studies of the forest transition in Vietnam, *Human ecology*.

Millennium Ecosystem Assessment, 2005, Ecosystems and human well-being: scenarios, Island Press, Washington D.C., USA.

Moran, E., Ojima, D., Buchmann, N., Canadell, J., Coomes, O. G., L., Jackson, R., Jaramillo, V., Laumann, G., Lavorel, S., Lambin, E., Leadley, P., Lourenço, N., Matson, P., McConnell, W. J., Morais, J., Murdiyarso, D., Pataki, D., Porter, J., P., A., Pitelka, L. F., Rajan, K., Ramankutty, N., Running, S., Stafford Smith, M., Turner II, B., Yagi, K., van der Leeuw, S., 2005, Global land project: Science plan and implementation strategy, IGBP Secretariat, Stockholm.

Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, D. R., Chan, K. M., Daily, G. C., Goldstein, J., Kareiva, P. M., Lonsdorf, E., Naidoo, R., Ricketts, T. H., Rebecca, M., 2009, Modeling multiple ecosystem services,biodiversity conservation, commodity production, and tradeoffs at landscape scales, *Frontiers in Ecology and Environment* **7**(1)**:**4-11.

Nicholson, E., Mace, G. M., Armsworth, P. R., Atkinson, G., Buckle, S., Clements, T., Ewers, R. M., Fa, J. E., Gardner, T. A., Gibbons, J., Grenyer, R., Metcalfe, R., Mourato, S., Muuls, M., Osborn, D., Reuman, D. C., Watson, C., Milner-Gulland, E. J., 2009, Priority research areas for ecosystem services in a changing world, *Journal of Applied Ecology* **46**(6)**:**1139-1144.

O'Farrell, P. J., Donaldson, J. S., Hoffman, M. T., 2007, The influence of ecosystem goods and services on livestock management practices on the Bokkeveld plateau, South Africa, *Agriculture, Ecosystems & Environment* **122**(3)**:**312-324.

Ostrom, E., 2009, A General Framework for Analyzing Sustainability of Social-Ecological Systems, *Science* **325**(5939)**:**419-422.

Ozenda, P., 1985, La végétation de la chaîne alpine dans l'espace montagnard européen, Masson, Paris, France. .

Pascual, U., Muradian, R., Brander, L., GÃ³mez-Baggethun, E., MartÃ-n-LÃ³pez, M., Verman, M., Armsworth, P., Christie, M., Cornelissen, H., Eppink, F., 2010, The economics of valuing ecosystem services and biodiversity, *TEEBâ€"Ecological and Economic Foundation*.

Pauli, H., Gottfried, M., Grabherr, G., 2003, Effect of climate change on the alpine and nival vegetation of the Alps, *J. Mt. Ecol.* **7 (Suppl.):**9-12.

Pereira, E., Queiroz, C., Pereira, H. M., Vicente, L., 2005, Ecosystem services and human-well-being: a participatory study in a mountain community in Portugal, *Ecology and society* **10**(2).

Perrings, C., Duraiappah, A., Larigauderie, A., Mooney, H., 2011, The Biodiversity and Ecosystem Services Science-Policy Interface, *Science* **331**(6021)**:**1139-1140.

Peterson, G. D., Cumming, G. S., Carpenter, S. R., 2003, Scenario planning: a tool for conservation in an uncertain world, *Conservation biology* **17**(2)**:**358-366.

Petz, K., van Oudenhoven, A. P. E., 2012, Modelling land management effect on ecosystem functions and services: a study in the Netherlands, *International Journal of Biodiversity Science, Ecosystem Services & Management***:**1-21.

Pieroni, A., Giusti, M., 2009, Alpine ethnobotany in Italy: traditional knowledge of gastronomic and medicinal plants among the Occitans of the upper Varaita valley, Piedmont, *Journal of Ethnobiology and Ethnomedicine* **5**(1)**:**32.

Quétier, 2006, Vulnérabilité des écosystèmes semi-naturels européens aux changements d'utilisations des terres, in: *Biologie des systèmes intégrés, Agronomie-Environnement*, Ecole supérieure Agronomique de Montpellier, Montpellier, pp. 269.

Quétier, F., Lavorel, S., Thuillier, W., Davies, I., 2007a, Plant-trait-based modelling assessment of ecosystem services sensitivity to land-use change, *Ecological Applications* **17**(8)**:**2377-2386.

Quétier, F., Liancourt, P., Thébault, A., Davies, I. D., Lavorel, S., 2012, Predicting past and present management effects on sub-alpine grasslands using plant traits, *Plant Ecology and Diversity* **in press**.

Quétier, F., Rivoal, F., Marty, P., de Chazal, J., Thuiller, W., Lavorel, S., 2010, Social representations of an alpine grassland landscape and socio-political discourses on rural development, *Regional Environmental Change* **10**(2)**:**119-130.

Quétier, F., Thebault, A., Lavorel, S., 2007b, Plant traits in a state and transition framework as markers of ecosystem response to land-use change, *Ecological Monographs* **77**(1)**:**33-52.

Raudsepp-Hearne, C., Peterson, G. D., Bennett, E. M., 2010, Ecosystem service bundles for analyzing tradeoffs in diverse landscapes, *Proceedings of the National Academy of Sciences* **107**(11)**:**5242-5247.

Ring, I., Hansjürgens, B., Elmqvist, T., Wittmer, H., Sukhdev, P., 2010, Challenges in framing the economics of ecosystems and biodiversity: the TEEB initiative, *Current opinion in Environmental Sustainability* **2:**1-12.

Robson, T. M., Baptist, F., Clement, J. C., Lavorel, S., 2010, Land use in subalpine grasslands affects nitrogen cycling via changes in plant community and soil microbial uptake dynamics, *Journal of Ecology* **98**(1)**:**62-73.

Rockstrom, J., Steffen, W., Noone, K., Persson, A., Chapin, F. S., Lambin, E., Lenton, T. M., Scheffer, M., Folke, C., Schellnhuber, H. J., Nykvist, B., de Wit, C. A., Hughes, T., van der Leeuw, S., Rodhe, H., Sorlin, S., Snyder, P. K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R. W., Fabry, V. J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., Foley, J., 2009, Planetary Boundaries: Exploring the Safe Operating Space for Humanity, *Ecology and Society* **14**(2).

Rodriguez, J. P., Beard, T. D., Bennett, E. M., Cumming, G. S., Cork, S. J., Agard, J., Dobson, A. P., Peterson, G. D., 2006, Trade-offs across space, time, and ecosystem services, *Ecology and society* **11**(1).

Rounsevell, M. D. A., Pedroli, B., Erb, K. H., Gramberger, M., Busck, A. G., Haberl, H., Kristensen, S., Kuemmerle, T., Lavorel, S., Lindner, M., Lotze-Campen, H., Metzger, M. J., Murray-Rust, D., Popp, A., Perez-Soba, M., Reenberge, A., Vadineanu, A., Verburg, P. H., Wolfslehner, B., 2012, Challenges for land system science, *Land Use Policy* **29**(4)**:**899-910.

Sala, O. E., Chapin, F. S., Armesto, J. J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L. F., Jackson, R. B., Kinzig, A., Leemans, R., Lodge, D. M., Mooney, H. A., Oesterheld, M., Poff, N. L., Sykes, M. T., Walker, B. H., Walker, M., Wall, D. H., 2000, Biodiversity - Global biodiversity scenarios for the year 2100, *Science* **287**(5459)**:**1770-1774.

Salles, J. M., 2011, Valuing biodiversity and ecosystem services: Why put economic values on Nature?, *Comptes Rendus Biologies* **334**(5-6)**:**469-482.

Schweitzer, J. A., Bailey, J. K., Fischer, D. G., LeRoy, C. J., Lonsdorf, E. V., Whitham, T. G., Hart, S. C., 2008, Plant-soil-microorganism interactions: heritable relationship between plant genotype and associated soil microorganisms, *Ecology* **89**(3)**:**773-781.

Seppelt, R., Dormann, C. F., Eppink, F. V., Lautenbach, S., Schmidt, S., 2011, A quantitative review of ecosystem service studies: approaches, shortcomings and the road ahead, *Journal of Applied Ecology* **48**(3)**:**630-636.

Simoncini, R., 2009, Developing an integrated approach to enhance the delivering of environmental goods and services by agro-ecosystems, *Regional Environmental Change* **9**(3)**:**153-167.

Soliva, R., 2007, Landscape stories: Using ideal type narratives as a heuristic device in rural studies, *Journal of Rural Studies* **23**(1)**:**62-74.

Steffen, W., Crutzen, P. J., McNeill, J. R., 2007, The Anthropocene: Are Humans Now Overwhelming the Great Forces of Nature, *AMBIO: A Journal of the Human Environment* **36**(8)**:**614-621.

Stevenson, R. J., 2011, A revised framework for coupled human and natural systems, propagating thresholds, and managing environmental problems, *Physics and Chemistry of the Earth* **36**(9-11)**:**342-351.

Strager, M. P., Rosenberger, R. S., 2006, Incorporating stakeholder preferences for land conservation: Weights and measures in spatial MCA, *Ecological Economics* **58**(1)**:**79-92.

Swetnam, R. D., Fisher, B., Mbilinyi, B. P., Munishi, P. K. T., Willcock, S., Ricketts, T., Mwakalila, S., Balmford, A., Burgess, N. D., Marshall, A. R., Lewis, S. L., 2011, Mapping socio-economic scenarios of land cover change: A GIS method to enable ecosystem service modelling, *Journal of Environmental Management* **92**(3)**:**563-574.

Swinton, S. M., Lupi, F., Robertson, P., Hamilto, S. K., 2007, Ecosystem services and agriculture: Cultivating agricultural ecosystems for diverse benefits, *Ecological Economics* **64:**245-252.

Tappeiner, U., Borsdorf, A., Tasser, E., 2008, Mapping the Alps. Society - Economy - Environment, Spektrum Akademischer Verlag.

Tappeiner, U. T., G.; Hilbert, A.; Mattanovich, E. , 2003, The EU Agricultural Policy and the Environment. Evaluation of the Alpine Region, Blackwell Wissenschafts-Verlag, Berlin (u.a.), pp. 275.

Tasser, E., Tappeiner, U., 2002, Impact of land use changes on mountain vegetation, *Applied Vegetation Science* **5**(2)**:**173-184.

Tress, G., Tress, B., Fry, G., 2005, Clarifying integrative research concepts in landscape ecology, *Landscape Ecology* **20**(4)**:**479-493.

UK National Ecosystem Assessment, 2011, The UK National Ecosystem Assessment: Synthesis of the Key Findings, UNEP-WCMC, Cambridge.

vanWey, L. K., Ostrom, E., Mertsky, V., 2005, Theories underlying the study of human-environment interactions, in: *Seeing the forest and the trees. Human-environment interactions in forest ecosystems* (E. F. Moran, E. Ostrom, eds.), Massachusetts Institute of Technology, Cambridge, pp. 23-56.

Veldkamp, A., 2009, Investigating land dynamics: future research perspectives, *Journal of Land Use Science* **4**(1-2)**:**5-14.

Vihervaara, P., Rönkä, M., Walls, M., 2010, Trends in Ecosystem Service Research: Early Steps and Current Drivers, *Ambio* **39**(4)**:**p. 314-324.

Vittoz, P., Selldorf, P., Eggenberg, S., Maire, S., 2005, Festuca paniculata meadows in Ticino (Switzerland) and their Alpine environment, *Botanica Helvetica* **115**(1)**:**33-48.

Vivant, L., 2007, Impact de l'évolution climatique sur les pratiques touristiques en milieu montagnard. Vallée de la Haute-Romanche, in: *Laboratoire de géographie alpine*, Université Joseph Fourier, Grenoble, pp. 46.

Wallace, K. J., 2007, Classification of ecosystem services: Problems and solutions, *Biological Conservation* **139**(3-4)**:**235-246.

Wardle, D., Barker, G., Bonner, K., Nicholson, K., 1998, Can comparative approaches based on plant ecophysiological traits predict the nature of biotic interactions and individual plant species effects in ecosystems?, *Journal of Ecology* **86**(3)**:**405-420.

Wardle, D. A., Bardgett, R. D., Klironomos, J. N., Setälä, H., Van Der Putten, W. H., Wall, D. H., 2004, Ecological linkages between aboveground and belowground biota, *Science* **304**(5677)**:**1629-1633.

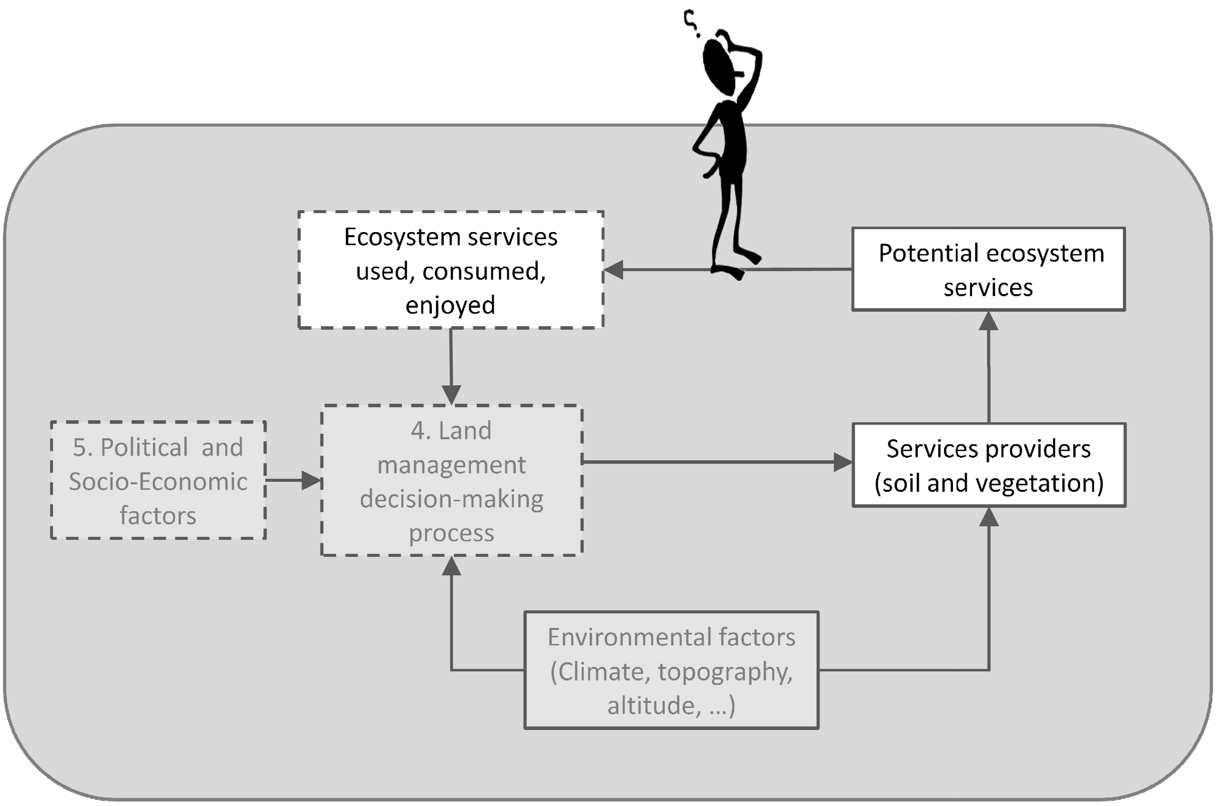
Westman, W. E., 1977, How much are nature's services worth? , *Science* **197:**960-964.

Reed, M. S., Graves, A., Dandy, N., Posthumus, H., Hubacek, K., Morris, J., Prell, C., Quinn, C. H., Stringer, L. C., 2009, Who's in and why? A typology of stakeholder analysis methods for natural resource management, *Journal of Environmental Management* **90**(5)**:**1933-1949.

Partie I

De l'écosystème à l'homme

Perceptions et modélisation des services des écosystèmes



Chapitre 2

Implications de la diversité de définitions des services écosystémiques[7]

Résumé

Le concept de service écosystémique est utilisé par de nombreuses disciplines scientifiques et commence à être largement utilisé dans le domaine politique et entrepreneurial. Pourtant plusieurs définitions et usages du concept coexistent, ainsi que des termes tels que services écologiques, environnementaux ou du paysage. Nous suggérons que cette variété terminologique traduit des différences de compréhension du concept. Celle-ci peut compliquer son utilisation pour la conservation de la nature et la gestion des ressources naturelles. Une application aux services fournis par des prairies semi-naturelles montre que ces différences peuvent amener à des évaluations très contrastées, que ce soit en termes de qualité, quantité ou localisation des services. Afin d'éviter ces problèmes un compromis doit être trouvé entre une définition élargie et utile pour la communication et les politiques à grande échelle et une définition plus précise et donc plus adaptée aux actions de gestion des écosystèmes et aux exigences d'une comptabilité nationale ou internationale des services.

Abstract

The ecosystem services concept is used in different scientific disciplines and is spreading into policy and business circles to draw attention to the benefits that people receive from biodiversity and ecosystems. Yet the concept remains multiform and is used interchangeably with a range of other terms such as ecological, landscape or environmental services. We argue that lexical differences in fact result from different understandings of the concept, which could slow its use in nature conservation or sustainable resource use. An application to semi-natural grasslands shows that such differences could lead to very different assessments, of quality, quantity and location of ecosystem services. We argue that a compromise must be found

[7] This chapter is published and can be cited as : Lamarque, P., Quetier, F., Lavorel, S., 2011, The diversity of the ecosystem services concept and its implications for their assessment and management, *Comptes Rendus Biologies* **334**(5-6)**:**441-449.

between a broad and simple definition, which is useful for communicating the concept and large-scale policies, and a more refined definition for research and implementation goals such as environmental management and national and international assessments and accounting.

1 Introduction

Although the deliberate identification of the range of goods and services that people obtain from nature (e.g. game, berries and fruit…) is not new it has received increasing attention in recent years under the banner of “nature’s services” or “ecosystem services” (Daily, 1997). This new way of framing the relationships between biodiversity, ecosystems and human well-being first gained strength in the field of nature conservation during the 1990s and later spread through a wide range of scientific disciplines (Fisher et al., 2009; Vihervaara et al., 2010) and more recently into policy and business circles (Millenium Ecosystem Assesment, 2005) (TEEB (The Economics of Ecosystems and Biodiversity), 2009).

The concept has provided a new, anthropocentric, justification for conserving species and ecosystems, based on our dependence on the goods and services they provide. Not only has it been widely used to draw attention to the importance of the benefits that people receive from biodiversity and ecosystems, it has also developed into a useful concept for framing the study of the relationships between nature, including both species and whole ecosystems, and the livelihoods of the communities that use or benefit from it. Part of the ecosystem and community ecology research communities took up the term as it shifted its focus from the effects of species number (e.g.(Balvanera et al., 2006)) on ecosystem functions such as productivity to the effects of the identity and abundance of species with particular sets of traits (i.e. functional diversity – (Diaz and Cabido, 2001)) on ecosystem services (e.g. (Diaz et al., 2007; Hooper et al., 2005)). Scientists working in the fields of agriculture, rangelands, forestry or natural resources in general have now taken up the concept of ecosystem services when referring to their positive outcomes for society, which were previously framed in terms of amenities or functions (as in multifunctional agriculture) (e.g. (Patterson and Coelho, 2009). These are used to better justify their practices or the considerable public support they sometimes receive (e.g. agri-environmental schemes under the European Union Common Agricultural Policy). The valuation of ecosystems by economists is not a new endeavour (e.g. (Krutilla, 1967; Krutilla and Fisher, 1975; Westman, 1977)) but its importance has grown considerably as market-based instruments have gained strength in the formulation and implementation of conservation policies worldwide (e.g. (Costanza et al., 1997; Ring et al., 2010)).

As the number of scientific disciplines that refer to the ecosystem services concept grows, and with its incorporation into political and corporate discourse, the concept is becoming multi-form and harder to grasp, and it has generated debates about definitions and classifications (e.g. (Boyd and Banzhaf, 2007; Costanza, 2008; Fisher and Turner, 2008; Fisher et al., 2009; Granek et al.; Wallace, 2007)). The aim of this paper is to highlight the implication of terminological diversity around the ecosystem services concept rather than open a semantic debate. We first review the general terminology that has gained currency in the environmental literature, with a specific focus on the diversity of meanings and approaches that have been applied for the use of the ecosystem services concept in the recent literature. We then briefly illustrate the implications of such definitional choices for a case study that aimed to quantify ecosystem services provided by mountain grasslands. We end with a discussion of the implications for scientific and operational purposes of the use of a diversity of definitions for the ecosystem services concept.

2 Terminological diversity in concepts of nature's services to society

2.1 The different broad terminologies of nature's services

While the main term used in the ecological and nature conservation literature to describe all things nature provide us is "ecosystem services", a series of related terms and concepts (merge here under a generic term "nature's services" (borrowed from (Daily, 1997)) have been developed in other contexts and disciplines.

Ecosystem services *sensu stricto* are broadly defined in the (Millennium Ecosystem Assessment, 2003) as the benefits people obtain from ecosystems (Table 1) and are classified in four categories: provisioning services (i.e. products obtained from ecosystems, such as food, fibre or timber), regulating services (e.g. flood or pest control and climate regulation), cultural services (i.e. non-material benefits such as aesthetic and recreational enjoyment) and supporting services (i.e. those services that are necessary for the proper delivery of the other three types of services, such as nutrient cycling). The validity of this last category has since been questioned as it amounts to mixing "ends" (i.e. services) and "means" (i.e. the ecological processes necessary) (Wallace, 2007). In a farming context, the concept of ecosystem services has also been used to refer to "input services" and "output services" for agriculture (Zhang et al., 2007). In addition, the term ecosystem goods (as in goods and services) is sometimes used for those services that have a direct market value such as food but both tangible goods and immaterial services provided by ecosystems are now generally labelled as ecosystem services.

Ecological services have been used by some authors as a synonym to ecosystem services (e.g. (Havstad et al., 2007; Moberg and Folke, 1999) but the term sometimes refers to services provided by a particular species or group of species rather than processes occurring at the ecosystem level (Losey and Vaughan, 2006).

Landscape services and the terms land, land-use and landscape functions are widely used when referring to services supplied by regions, landscapes or land-use systems with the technical and socio-economic characteristics of the land-use system being taken into consideration together with abiotic and biotic components (e.g. (Verburg et al., 2009; Willemen et al., 2008)). Landscape functions are often considered in terms of their “potential” for human use (e.g. (Bastian, 2000)). Other authors suggest that landscape services differ from ecosystem services in that they take explicitly into account the underlying role of spatial patterns, landscape elements and horizontal landscape processes (following (Termorshuizen and Opdam, 2009)).

Environmental services are often used as a synonym of ecosystem services in PES schemes (Payment for Environmental Services), where stewards are paid by third party beneficiaries for an activity aimed at intentionally transforming or maintaining some useful characteristics of an ecosystem (or landscape) (Aznar, 2002). Other authors have proposed to use the term environmental services to label human-made services which totally or partially substitute ecosystem services (Koellner and Grêt-Regamey, Unpublished). This fits with the use of the term to label waste and water management services (as in the case of the company Veolia Environnement© which claims to be a “world leader in environmental services”). The term sometimes also refers to the services provided by the abiotic environment such as the wind or water regimes used for generating electricity. In this case, the links with fauna and flora (i.e. biodiversity) are indirect (Haines-Young and Potschin, 2010).

These terminologies (Figure 1) differ in terms of (1) the key components and processes necessary to deliver an ecosystem service, called hereafter “services providers” (this term the “service-providing units” (Luck et al., 2003) with the “ecosystem services providers” (Kremen, 2005)), ranging from species to landscapes, and (2) human interventions in their delivery. Figure 1 illustrates how a shift in focus from specific biotic components of ecosystems to whole ecosystems, landscapes and finally man-made substitutes leads to a decrease in the importance of biodiversity to the provision of services. In parallel, the ecological knowledge required to understand the role of biodiversity decreases, whereas knowledge on human processes is increasingly incorporated.

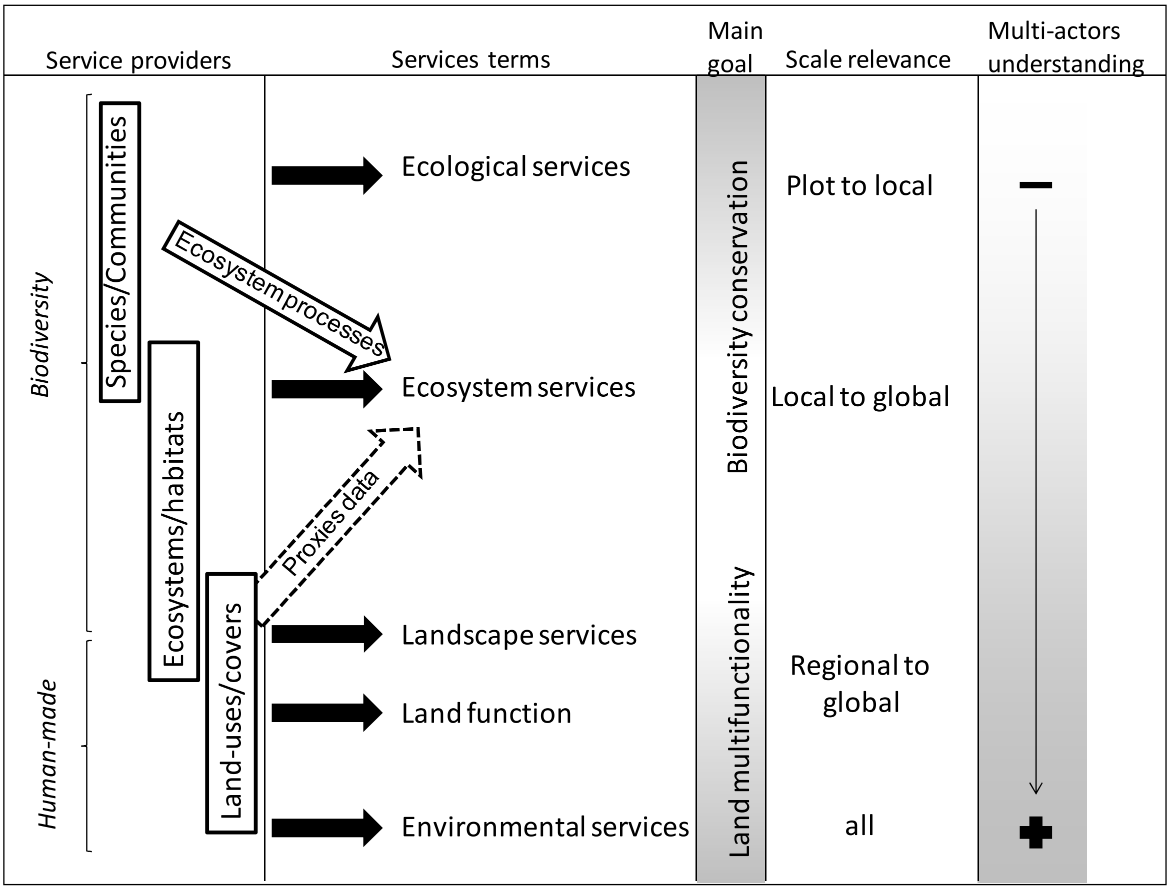
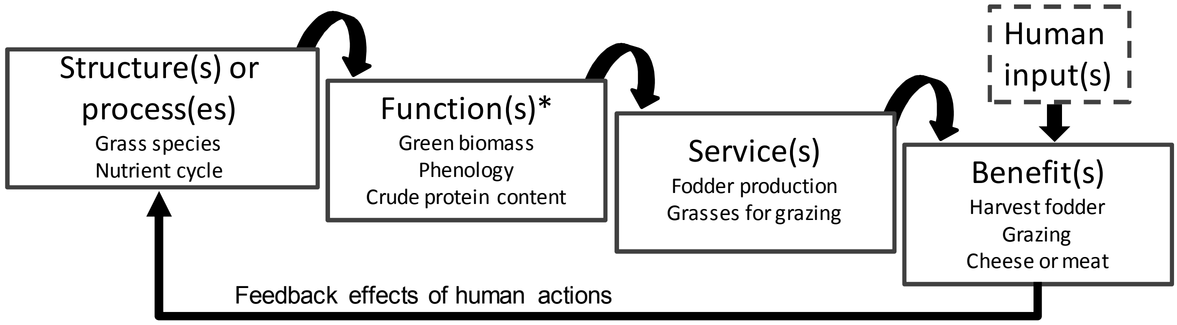


Figure 1: Differences between terms used to describe nature's services in their services providers units, management goals, scales of relevance and consideration for multiple actors.

2.2 Diversity of ecosystem services definitions

We analysed how the most common and contrasted definitions of the term "ecosystem services" are distributed along a cascade of contributing elements that helps to dissect the ecosystem services concept into "functions", "services" and "benefits", as proposed by Haines-Young and Potschin (Haines-Young and Potschin, 2009, in press) and (de Groot et al., 2010) and illustrated in figure 2. The different definitions are summarized in Table 1.



* subset of biophysical structure or process providing the service

Figure 2: Conceptual cascade of ecosystem services from processes to benefits (adapted from (Haines-Young and Potschin, 2010) and (de Groot et al., 2010)) and example of fodder production in mountain grasslands.

In this cascade, structure and process are the biophysical components (e.g. species and their abundance) and processes (e.g. interactions between species and ecosystem compartments) which underpin the potentiality for the ecosystem to deliver one or multiple services. This potentiality is referred to as ecosystem functions. Functions are themselves translated into services when they are used, consumed or enjoyed by humans ((Fisher et al., 2009). This makes ecosystem services location (e.g. avalanche regulation is only relevant if there are people living downhill) and time-dependent as well as beneficiary-dependent (different individuals or collectives benefit from different services (Diaz et al., 2006)). Finally because many benefits are in fact obtained by combining natural and human capital it can be useful to distinguish benefits from their strictly ecological inputs (Boyd and Banzhaf, 2007).

The main definitions considered here are not all located at the same place along the conceptual cascade (Figure 2) and the same terms can be used to describe different steps. Generally, confusion occurs at four critical points:

(1) Defining function and services; the confusion between services and function comes from the use of the word ‘function’ to describe the functioning of ecosystems (Jax, 2005) which is sometimes used as a synonymous of ecosystem properties (Boyd and Banzhaf, 2007; Costanza et al., 1997; Fisher et al., 2009; Wallace, 2007). Of the many structures and processes that occur within an ecosystem, not all are relevant to a particular service. For example, the avalanches protection service depends on the function of snow retention by trees, which depends on forest structure.

(2) Identifying the structure(s) or process(es) which allow the delivery of services: these ecosystem properties can be species, communities or ecological structures (e.g. layers in a forest or length of hedges) as well as complex cycle processes or fluxes or a combination of all the former. For example, through photosynthesis, a forest might provide a global service of carbon sequestration, and some species in the forest can be used for firewood while others are used for ornamental woodwork. Some ecosystem properties are affected by the ecosystem’s location in

the physical and ecological landscape (such as water flow, which also results from the ecosystem's location).

(3) Defining services and benefits: (Costanza et al., 1997; Millenium Ecosystem Assesment, 2005) define services as benefits while more recent papers separate benefits and ecosystem services, considering that the former are a product of the latter and other forms of capital (Boyd and Banzhaf, 2007; Fisher et al., 2009). The distinction is important in order to avoid double-counting in monetary valuation. For example, clean drinking water for consumption is a benefit dependant on a range of intermediate services such as clean water provision and processes such as nutrient cycling but the contribution of these intermediate services is already encompassed in that of the water.

(4) Defining direct vs. indirect provision. (Boyd and Banzhaf, 2007; Wallace, 2007) stressed the importance of taking into account services only directly used or consumed by humans. In response to their paper (Fisher and Turner, 2008; Fisher et al., 2009) pointed out that ecosystem services do not have to be utilized directly because "as long as human welfare is affected by ecological processes or functions they are services. This raises the question of differentiating "intermediate services" from the services that directly benefit individuals or collectives (see (Boyd and Banzhaf, 2007; Fisher and Turner, 2008)).

Authors	**Ecosystem components or processes**	**Function(s)**	**Service(s)**	**Benefit(s)**
	Services providers	**Potential services**	**Services used, consumed or enjoyed by human beneficiaries**	**Benefits obtain from ecosystem services and/or human-made services which improve human well-being**
de Groot et al, 2002, Teeb 2009, Haines-Young & Potschin (2010)	Ecological structure, habitat, ecosystem properties and supporting services	The potential that ecosystems have to deliver a service. 'Things' needed to deliver services	the direct and indirect contributions of ecosystem to human well being	Welfare gains generate
MA, 2005	Supporting services		Benefits people obtain from ecosystem	
Daily (ed), 1997	Complex natural cycle		The conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life	
Costanza et al, 1997	Functions : Refer variously to the habitat, biological system properties or processes of ecosystems		Services : the benefits human populations derive, directly or indirectly, from ecosystem functions	Human welfare are the results of ecosystem services [...] from natural capital stocks combined with manufactured and human capital services
Wallace, 2007	Ecosystem function as a synonym of ecosystem processes which are the complex interactions among biotic and abiotic elements of ecosystems that lead to a definite result. For urban or rural system with few natural elements could also be cultural elements of ecosystem or some combination with natural elements.		point where ecosystem direclty provides an asset that is used/consumed by one or more humans.	
Boyd and Banzhaf, 2007	Function and processes are intermediate to the production of final ecosystem services. Intermediate services		Final ES are components of nature, directly enjoyed, consumed, or used to yield human well-being. Ecological things or caractheristic, not process or function.	Use both ecological services and conventional goods and services (man-made input)
Fisher et al, 2009	Ecosystem structure and processes provide services		The aspects of ecosystems utilized (actively or passively) to produce human well being. Ecological phenomena do not have to be directly utilized	Point at which human welfare is directly affected and the point where other forms of capital (built, human, social) are likely needed to realize the gain in welfare
Diaz et al, 2006 and 2007	Ecosystem processes : Intrinsic processes and fluxes whereby an ecosystem maintains its integrity	Relevant ecosystem properties to ecosystem services	Benefits provided by ecosystems to humans	Ecosystem services contribute to making human life both possible and worth living
Kremen, 2005	Ecosystem services providers (species or entities)	Ecosystem services are the set of ecosystem functions that are ueful to humans		
Hooper et al, 2005	Various pools and fluxes	Ecosystem goods and services are the subset of function of utilitarian value to human		
Termorshuizen and Opdam, 2009	Landscape : spatial human-ecological system. Interaction between physical structure and human actions	Functions are translated into services when they are valued by people	Uses and values of landscape by people	
Willemen et al, 2008 et 2009	Socio-economic and biophysical variation of the landscape and the spatial and temporal interactions between the different components of the landscape	Landscape function : capacity of a landscape to provide goods and services to society.	Landscape services	
Verburg et al, 2009	Land use systems and ecosystems within the landscape	Land function : goods and services provided		

Table 1: Inventory of main definitions of ecosystem services, functions and benefits used in scientific literature and difference of interpretation in the framework of the cascade. Column headings follow the different boxes of Figure 2.

3 Case study

To discuss how the different definitions reviewed in Table 1 apply in a real world situation, we have applied them to the identification, quantification and mapping of ecosystem services provided by a 1300 ha area of mountain grasslands in the French Alps. The grasslands are mainly managed for livestock and the sake of illustration we focus our discussion on fodder production (see figure 2 for examples), although tourism and nature conservation are important activities in the area as well, each relating to an additional set of ecosystem services.

Before exploring how the definitions reviewed in Table 1 would label these different steps in the cascade we first focus on how local stakeholders understand the ecosystem services concept and map the different steps mentioned above along the cascade. This is important as ecosystem service scientists need to consult with beneficiaries of these services to establish the links between ecosystem functions and benefits (Fisher et al., 2009; Termorshuizen and Opdam, 2009). Stakeholders also expect scientists to produce results such as quantitative data and maps of ecosystem services that are framed according to their own terminology and needs.

3.1 Stakeholders understanding and expectations

We conducted semi-guided interviews on ecosystem services and biodiversity with 13 professionals of agriculture, nature conservation, tourism and rural development working in regional-level public administrations and NGOs and with 6 inhabitants (including farmers) of the case study area. Interviewees were asked about their knowledge and understanding of the notion of “ecosystem services”, and in the case of regional professional of the term itself. Only half of the regional professionals had heard about the concept, and half of these defined it correctly. Some of those who had heard of the term confused it with the broader concepts of amenities or agricultural multifunctionality on the basis of which farmers are subsidized. The representations by stakeholders of the different steps of the conceptual cascade for fodder production are shown in Table 2. Other ecosystem services identified by interviewees are in fact benefits as they imply labour or technical skills. They include honey production, ski resort and related recreation or job opportunities. Thus, although their understanding of the term “ecosystem services” was imprecise, interviewees acknowledged linkages between biodiversity and benefits they obtain from the area with economic or other values.

Interestingly, when identifying services delivered by grasslands interviewees tried to compare them to other ecosystem types such as forests in terms of their importance for the delivery of each service. They stated for example that well-managed grasslands can decrease the risk of avalanches but that this service is better supplied by a forest. This highlights people’s general knowledge of landscapes and ecosystem types. They have less insight into the particular processes that generate services within a given ecosystem. This could make the concept of ecosystem service harder to grasp than that of “landscape services” (see Figure 1).

Stakeholders interested in implementing the concept and using it as a communication tool (i.e. for providing multiple arguments to conserve biodiversity) mentioned the need for precise identifications, measurements and maps of the services in their area. Obtaining such data requires first that they know what it is they want to map: functions, services or benefits.

<table>
<tr><th></th><th>Structure (s) and process(es)</th><th>Function(s)</th><th>Services(s)</th><th>Benefit(s)</th></tr>
<tr><td>Maps of figure 3</td><td></td><td>Map a</td><td>Map b</td><td>Map c and d</td></tr>
<tr><td rowspan="2">Terms mentioned by interviewees</td><td colspan="2" rowspan="2">Biodiversity, soil fertility, soil genesis
Water availability
Generally not precise but people say that some component or process are needed to deliver ecosystem services</td><td>Fodder (quality and/or quantity)</td><td>Cheese</td></tr>
<tr><td colspan="2">Fodder for cattle, animal production, cheese, meat</td></tr>
</table>

Table 2: Fodder production services illustrated by stakeholder’s identification of ecosystem services delivered by grasslands (terms mentioned by interviewees to describe ecosystem services).

3.2 Implication for quantifying and then mapping ecosystem services

In order to illustrate the implication of the different definitions, four different variables around fodder production, each corresponding to different steps in the conceptual diagram (Figure 2), are used to quantify and then map ecosystem functions and services (Figure 3). The methodology is explained briefly below. Maps at the landscape scale are obtained by extrapolating data collected in 57 plots representing 8 different land-uses. For further details on data and mapping methodologies see (Lavorel et al., submitted).

Ecosystem services were identified on the basis of interviews with local farmers on their need and uses of grasslands. For them a good meadow for mowing or grazing (i.e. fodder production services) is the result of quality and quantity of grasses that corresponds to a combination of different ecosystem functions : grass quantity, quality and flowering phenology (Quétier et al., 2007a). Those functions are translated by researchers into measurable indicators such as annual green biomass production ($g.m^{-2}$) to evaluate grass quantity (Figure 3 - map a). Because some authors, especially in the ecological literature, confuse ecosystem functions and services, these maps could be considered by some as ecosystem services maps (see Table 1 for examples of such definitions of ecosystem services). However, a simple visual observation between green biomass ecosystem function (Figure 3 - map a) and agronomic quality ecosystem service (Figure 3 – map b) shows a different spatial pattern. For example, zone 1 (Figure 3) corresponding to summer meadows have low biomass production but have high quality and zone 2 (Figure 3) related to mown and fertilized grasslands are mainly valued for the biomass production and their early flowering. Hence, the area assessed as providing a high agronomic service is the same as the area with high biomass production. These differences illustrate the important distinction between function which can be delivered by several grasslands around the world and service which include the manner grasslands are used in a given farming system and which do it context-dependent (eg. culture, socio-economic). Finally, biological or ecological data are not sufficient to quantify or map benefits. These require agronomic, social or economic information on other inputs (which are anthropogenic rather than provided by ecosystems). Fodder benefits farmers as it is incorporated into the farming system (i.e. used to feed their flocks which then produce meat and thereby contribute to their livelihoods). It also benefits consumers outside of the local area in the form of food such as cheese or meat. Here we mapped benefits to cattle production using data of the number of days of livestock units / ha aggregated by land use based on farmers' interviews data. Map (c) in figure 3 shows that cattle grazing is on grasslands of high agronomic quality only for a few months on summer meadows (Figure 3-zone 1), and rather more concentrated and longer on areas close to farm buildings used for spring and autumn pastures. In the same way, a map of harvested fodder for hay (d) shows that topography and access constrain farmers to not always harvest the more productive areas. Note, that depending on the production system cattle grazing or harvested fodder are only one dimension of the benefit and are in this type of mountain system combined to determine the total benefit of grasslands to farmers. In spite of this, these last two maps demonstrate that ecosystem functions are not sufficient to assess benefits, additional human inputs such as labour, machinery (tractors), infrastructure (roads) and so on are needed.

All those differences between maps demonstrate that ecosystem function, services and benefits are not equivalent (Figure 3). Hence, this makes comparative studies (eg. between ecosystem services deliver by mountain grasslands sites) difficult.

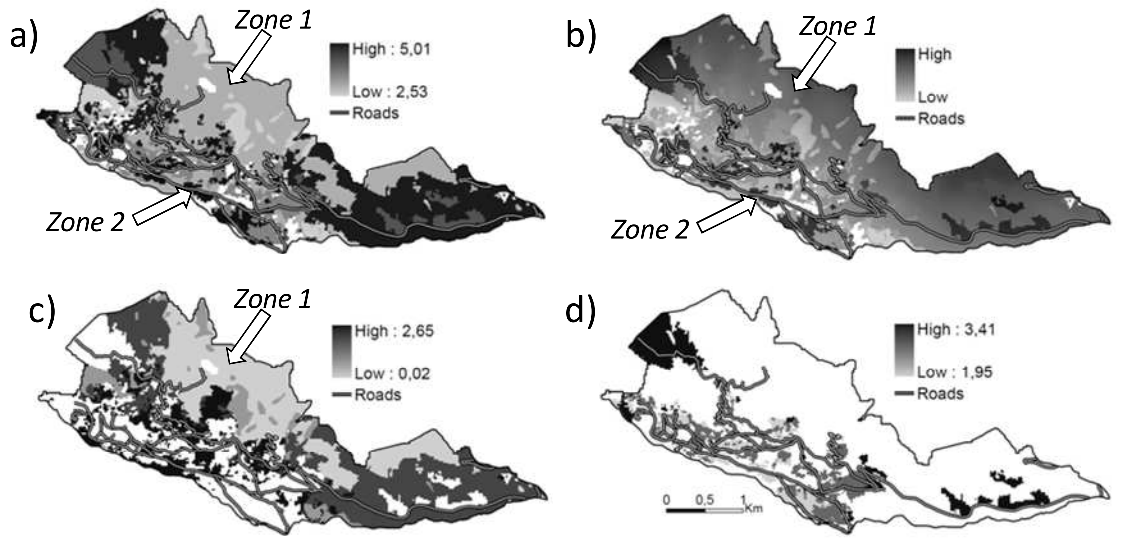


Figure 3 : Maps of nature's services related to fodder production according to different definitions: (a) Ecosystem function: Green biomass (tons/ha) (b) ecosystem services: agronomic value (unitless) obtained from a combination of different functions (green biomass, digestibility and phenology), and benefits : (c) the number of days of livestock units / ha and (d) hay production (tons/ha). Roads and tracks are added on maps as they are important elements of analysis.

4 Discussion

We subscribe to the idea that definitions and classifications of ecosystem services are purpose-dependent and should be judged on their usefulness for a particular purpose (eg. (Zhang et al., 2007)). Yet our review also shows that the coexistence of different terminologies and definitions could impede the on-the-ground use of the concept because of the difficulties in translating it into tangible, manageable, "things" to measure, count, qualify or map.

4.1 Precision and broadness of nature's services and associated definitions

There are advantages and drawbacks to a precise definition that distinguishes each step of the conceptual cascade relative to a broader definition that does not make these distinctions. The case study shows that different definitions lead to different spatial patterns of delivery. This would lead to divergent identification of areas with high ecosystem service delivery, with possible consequences for management choices or payments for ecosystem services. Therefore, the specifications of definitions for ecosystem services can have strong implications in the context of biodiversity conservation, the sustainable use of natural resources, or even rural development where site managers and decision-makers are expect concrete, practical and precise data on which to base their decisions. Hence, the distinction is useful for scientists aiming to quantify and then compare services. It is also useful in helping scientists clarify the needs and expectations of stakeholders in a context of increasingly participatory research in natural resource and ecosystem management (Barreteau et al., 2010b). We showed that differences between definitions are important sources of differences but we did not address differences within definitions due for example to quantification methods such as the use of proxy versus field data which can lead to inaccurate maps (Eigenbrod et al., 2010).

Building on-the-ground assessments on the basis of a broad definition can also lead to misunderstandings or departures from the original concept (Haines-Young et al., 2009) that could be amplified through dissemination and gradual transformation of original definitions. Yet, a precise and complex framework is inappropriate as a communication tool. A broader definition, carrying the main message that nature is useful to humans, is probably more appropriate for the general public and higher-level policy makers. It has in fact contributed to the concept's success (Vihervaara et al.). A simple definition also has the benefit of matching peoples' definitions and understanding (table 2). It is important to communicate on both services providers and ecosystem services in order to increase public awareness on the dependence of services upon ecosystem processes and components such as biodiversity in order to adopt sustainable management of ecosystem services.

We conclude that the full distinction between the four components of the conceptual cascade (Figure 2) is useful for the quantification of ecosystem services, for mapping or valuing for example, but that for general public communication purposes, a simplified distinction that merges service providers units and functions on the one hand, and services and benefits on the other should be sufficient.

4.2 Two contrasted but complementary uses of services.

According to our analysis of uses and definitions of nature's services by different authors, it appears that the term has two audiences. The concept is used as a tool for natural resources management or biodiversity conservation (see (Egoh et al., 2007) for a review) by those who adopt an anthropocentric and utilitarian approach. This approach is distinct from the intrinsic value pleading for the inherent worth of biodiversity, independent of its value to anyone or anything else (Eldredge, 2002). A second use of the term is that observed in land use planning studies (eg. (Grêt-Regamey et al., 2008)) where nature's services are used in a holistic approach centred around the conciliation of different human activities with environmental constraints and biodiversity. In the context of agroecosystems, this approach is consistent with others such as agricultural or landscape multifunctionality which suggest that "agriculture can provide numerous commodity and non-commodity outputs, some of which benefit the public without compensating the farmer" (Lovell et al.). Some authors consider not only nature's services from agriculture but also to agriculture (Zhang et al., 2007).

Faced with these two audiences, two solutions emerge: keeping a common term and accepting ambiguity or using two different terminologies. We propose "ecosystem services" for use in the context of biodiversity conservation and natural ecosystems, both because it was its original goal and because most ecosystem services depend on biodiversity components, and the term “landscape services” for use in land-use planning, because it is based on land-use patterns and practices and it is open to human inputs (labour, technology etc.).

5 Conclusion

By accepting that the value of biodiversity and ecosystems be weighted against other components of human well-being, in particular using the tools of economic analysis, conservationists have opened the door to closer cooperation with policy makers and business circles (Ring et al., 2010), in keeping with their objectives of “mainstreaming conservation into the everyday decisions of the business and public sectors” (Balmford and Cowling, 2006). However, the complete consequences of this shift in vocabulary and in the underlying sets of values, from bio- or eco-centric to anthropocentric and utilitarian justifications for the conservation of wild nature, are yet to be revealed. Ecosystem services and more broadly nature’s services is one tool among many to communicate and justify biodiversity conservation. But as this paper demonstrates, this concept is difficult to grasp. The concept’s integrative and federating approach is appealing and helps translate complex ecological processes into a common and simple vocabulary understandable in multidisciplinary scientific and political discourses (Vihervaara et al.), yet it has also become important to move towards more precise definitions of what ecosystems services are, not only for effective implementation and use, but also to avoid misrepresentations which could undermine the credibility of the concept.

Acknowledgements

We thank Marie-Pascale Colace, Karl Grigulis, Audrey Orcel, Stéphanie Périquet, Marion Salvi, Francesco de Bello and Eric Deboeuf for data collection, the Ecrins National Park for support, Villar d’Arène farmers for access to their land, all those we interviewed for the time they took to answer our questions, and the Joseph Fourier Alpine Station for logistical support. This research was conducted on the long term research site “Zone Atelier Alpes”, a member of the ILTER-Europe network. ZAA publication n°xxxx. This work is supported by ANR through the FP6 BiodivERsA Eranet VITAL project.

References

Aznar, O., 2002, Services environnementaux et espaces ruraux. Une approche par l'économie des services., in: *Sciences économiques*, Université de bourgogne, pp. 275.

Balmford, A., Cowling, R. M., 2006, Fusion or failure? The future of conservation biology, *Conservation Biology* **20**(3)**:**692-695.

Balvanera, P., Pfisterer, A. B., Buchmann, N., He, J. S., Nakashizuka, T., Raffaelli, D., Schmid, B., 2006, Quantifying the evidence for biodiversity effects on ecosystem functioning and services, *Ecology Letters* **9**(10)**:**1146-1156.

Barreteau, O., Bots, P. W. G., Daniell, K. A., 2010, A framework for clarifying "Participation" in participatory research to prevent its rejection for the wrong reasons, *Ecology and Society* **15 (2)**(1).

Bastian, O., 2000, Landscape classification in Saxony (Germany) - a tool for holistic regional planning, *Landscape and Urban Planning* **50**(1-3)**:**145-155.

Boyd, J., Banzhaf, S., 2007, What are ecosystem services? The need for standardized environmental accounting units, *Ecological Economics* **63**(2-3)**:**616-626.

Costanza, R., 2008, Ecosystem services: Multiple classification systems are needed, *Biological Conservation* **141**(2)**:**350-352.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R. V., Paruelo, J., Raskin, R. G., Sutton, P., van den Belt, M., 1997, The value of the worl's ecosystem services and natural capital, *Nature* **387:**253-260.

Daily, G. C., 1997, Nature's Services: Societal Dependence on Natural Ecosystems, Island Press, Washington D.C.

de Groot, R.S., M.A. Wilson, and R.M.J. Boumans, 2002. *A typology for the classification, description and valuation of ecosystem functions, goods and services.* Ecological Economics,. **41**(3): p. 393-408.

de Groot, R. S., Alkemade, R., Braat, L., Hein, L., Willemen, L., 2010, Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making, *Ecological Complexity* **7**(3)**:**260-272.

Diaz, S., Cabido, M., 2001, Vive la difference: plant functional diversity matters to ecosystem processes, *Trends in Ecology & Evolution* **16**(11)**:**646-655.

Diaz, S., Fargione, J., Stuart Chapin, F., Tilman, D., 2006, Biodiversity Loss Threatens Human Well-Being, *PLoS Biology* **4**(8)**:**1300-1305.

Diaz, S., Lavorel, S., de Bello, F., Quétier, F., Grigulis, K., Robson, T. M., 2007, Incorporating plant functional diversity effects in ecosystem service assessments, *Proceedings of the National Academy of Science* **104**(52)**:**20684-20689.

Egoh, B., Rouget, M., Reyers, B., Knight, A. T., Cowling, R. M., van Jaarsveld, A. S., Welz, A., 2007, Integrating ecosystem services into conservation assessments: A review, *Ecological Economics* **63**(4)**:**714-721.

Eigenbrod, F., Armsworth, P. R., Anderson, B. J., Heinemeyer, A., Gillings, S., Roy, D. B., Thomas, C. D., Gaston, K. J., 2010, The impact of proxy-based methods on mapping the distribution of ecosystem services, *Journal of Applied Ecology* **47**(2)**:**377-385.

Eldredge, N., 2002, Life on earth. An encyclopedia of biodiversity, ecologhy, and evolution, ABC-Clio, Santa Barbara, California.

Fisher, B., Turner, R. K., 2008, Ecosystem services: Classification for valuation, *Biological Conservation* **141**(5)**:**1167-1169.

Fisher, B., Turner, R. K., Morling, P., 2009, Defining and classifying ecosystem services for decision making, *Ecological Economics* **68**(3)**:**643-653.

Granek, E. F., Polasky, S., Kappel, C. V., Reed, D. J., Stoms, D. M., Koch, E. W., Kennedy, C. J., Cramer, L. A., Hacker, S. D., Barbier, E. B., Aswani, S., Ruckelshaus, M., Perillo, G. M. E., Silliman, B. R., Muthiga, N., Bael, D., Wolanski, E., 2010, Ecosystem Services as a Common Language for Coastal Ecosystem-Based Management, *Conservation Biology* **24**(1)**:**207-216.

Grêt-Regamey, A., Bebi, P., Bishop, I. D., Schmid, W. A., 2008, Linking GIS-based models to value ecosystem services in an Alpine region, *Journal of Environmental Management* **89**(3)**:**197-208.

Haines-Young, R., Potschin, M., 2009, in press, The links between biodiversity, ecosystem services and human well-being, in: *Ecosystem ecology: A New Synthesis* (D. Raffaelli, C. Frid, eds.), CUP, Cambridge.

Haines-Young, R., Potschin, M., 2010, The links between biodiversity, ecosystem services and human well-being, in: *Ecosystem ecology: A New Synthesis* (D. Raffaelli, C. Frid, eds.), CUP, Cambridge.

Haines-Young, R., Potschin, M., Groot, d. R. S., Kienmast, F., Bolliger, J., 2009, Towards a Common International Classification of Ecosystem services (CICES)) for Integrated Environmental Common International Classification of Ecosystem services (CICES)) for Integrated Environmental and Economic Accounting.

Havstad, K. M., Peters, D. P. C., Skaggs, R., Brown, J., Bestelmeyer, B., Fredrickson, E., Herrick, J., Wright, J., 2007, Ecological services to and from rangelands of the United States, *Ecological Economics* **64**(2)**:**261-268.

Hooper, D. U., Chapin, F. S., Ewel, J. J., Hector, A., Inchausti, P., Lavorel, S., Lawton, J. H., Lodge, D. M., Loreau, M., Naeem, S., Schmid, B., Setala, H., Symstad, A. J., Vandermeer, J., Wardle, D. A., 2005, Effects of biodiversity on ecosystem functioning: A consensus of current knowledge, *Ecological Monographs* **75**(1)**:**3-35.

Jax, K., 2005, Function and "functioning" in ecology: what does it mean?, *Oikos* **111**(3)**:**641-648.

Koellner, T., Grêt-Regamey, A., Unpublished, Ecosystem services in green national accounting, in: *Manuscript in preparation for Ecological Economics.*

Kremen, C., 2005, Managing ecosystem services: what do we need to know about their ecology?, *Ecology Letters* **8:**468-479.

Krutilla, J. V., 1967, Conservation reconsidered, *American Economic Review* **57:**777-786.

Krutilla, J. V., Fisher, A. C., 1975, The economics of natural environments - Studies in the valuation of commodity and amenity resources, in: *Resources for the Future*, Washington, D.C., USA.

Lavorel, S., Grigulis, K., Lamarque, P., Colace, M.-P., Garden, D., Girel, J., Pellet, G., Rome, M., Douzet, R., submitted, Using plant functional traits to understand the landscape distribution of multiple ecosystem services, *Manuscript submitted to Journal of ecology*.

Losey, J. E., Vaughan, M., 2006, The economic value of ecological services provided by insects, *Bioscience* **56**(4)**:**311-323.

Lovell, S. T., DeSantis, S. r., Nathan, C. A., Olson, M. B., Ernesto Méndez, V., Kominami, H. C., Erickson, D. L., Morris, K. S., Morris, W. B., 2010, Integrating agroecology and landscape multifunctionality in Vermont: An evolving framework to evaluate the design of agroecosystems, *Agricultural Systems* **103**(5)**:**327-341.

Luck, G. W., Daily, G. C., Ehrlich, P. R., 2003, Population diversity and ecosystem services, *Trends in Ecology & Evolution* **18**(7)**:**331-336.

Millenium Ecosystem Assesment, 2005, Ecosystem and Human Well-being: Synthesis (I. Press, ed.), Washington DC.

Millennium Ecosystem Assessment, 2003, Millennium Ecosystem Assessment, Ecosystems and Human Well-being: A Framework for Assessment, World Resources Institute, Washington, DC.

Moberg, F., Folke, C., 1999, Ecological goods and services of coral reef ecosystems, *Ecological Economics* **29**(2)**:**215-233.

Patterson, T. M., Coelho, D. L., 2009, Ecosystem services: Foundations, opportunities, and challenges for the forest products sector, *Forest Ecology and Management* **257:**1637-1646.

Quétier, F., Lavorel, S., Thuillier, W., Davies, I., 2007, Plant-trait-based modelling assessment of ecosystem services sensitivity to land-use change, *Ecological Applications* **17**(8)**:**2377-2386.

Ring, I., Hansjürgens, B., Elmqvist, T., Wittmer, H., Sukhdev, P., 2010, Challenges in framing the economics of ecosystems and biodiversity: the TEEB initiative, *Current opinion in Environmental Sustainability* **2:**1-12.

TEEB (The Economics of Ecosystems and Biodiversity), 2009, TEEB for Policy Makers Summary: Responding to the Value of Nature, Available at: www.teebweb.org, accessed 10 june 2010.

Termorshuizen, J. W., Opdam, P., 2009, Landscape services as a bridge between landscape ecology and sustainable development *Landscape Ecology* **Volume 24**(Number 8)**:**1037-1052.

Verburg, P. H., van de Steeg, J., Veldkamp, A., Willemen, L., 2009, From land cover change to land function dynamics: A major challenge to improve land characterization, *Journal of Environmental Management* **90**(3)**:**1327-1335.

Vihervaara, P., Rönkä, M., Walls, M., 2010, Trends in Ecosystem Service Research: Early Steps and Current Drivers, *Ambio* **39**(4)**:**p. 314-324.

Wallace, K. J., 2007, Classification of ecosystem services: Problems and solutions, *Biological Conservation* **139**(3-4)**:**235-246.

Westman, W. E., 1977, How much are nature's services worth? , *Science* **197:**960-964.

Willemen, L., Verburg, P. H., Hein, L., van Mensvoort, M. E. F., 2008, Spatial characterization of landscape functions, *Landscape and Urban Planning* **88:**34-43.

Zhang, W., Ricketts, T. H., Kremen, C., Carney, K., Swinton, S. M., 2007, Ecosystem services and dis-services to agriculture, *Ecological Economics* **64:**253-260.

Chapitre 3

Perception des services par les acteurs locaux[8]

Abstract

The concept of ecosystem services is increasingly being used by scientists and policy makers. However, most studies in this area have focussed on factors that regulate ecosystem functions (i.e. the potential to deliver ecosystem services) or the supply of ecosystem services. In contrast, demand for ecosystem services (i.e. the needs of beneficiaries) or understanding of the concept and the relative ranking of different ecosystem services by beneficiaries has received limited attention. The aim of this study was to identify in three European mountain regions the ecosystem services of grassland that different stakeholders identify (which ecosystem services for whom), the relative rankings of these ecosystem services, and how stakeholders perceive the provision of these ecosystem services to be related to agricultural activities. We found differences: (1) between farmers' perceptions of ecosystem services across regions; and (2) within regions, between knowledge of ecosystem services gained by regional experts through education and farmers' local field-based knowledge. Nevertheless, we identified a common set of ecosystem services that were considered important by stakeholders across the three regions, including soil stability, water quantity and quality, forage quality, conservation of botanical diversity, aesthetics and recreation (for regional experts), and forage quantity and aesthetic (for local farmers). We observed two contrasting stakeholder representations of the effects of agricultural management on ecosystem services delivery, one negative and the other positive (considering low to medium management intensity). These representations were determined by stakeholders' perceptions of the relationships between soil fertility and biodiversity. Overall, differences in perceptions highlighted in this study show that practitioners, policy makers and researchers should be more explicit in their uses of the ecosystem services concept in order to be correctly understood and to foster improved communication among stakeholders.

[8] This chapter is published and can be cited as : Lamarque, P., Tappeiner, U., Turner, C., Steinbacher, M., Bardgett, R. D., Szukics, U., Schermer, M., Lavorel, S., 2011, Stakeholder perceptions of grassland ecosystem services in relation to knowledge on soil fertility and biodiversity, *Regional Environmental Change* **11**(4)**:**791-804.

Remarque : Les enquêtes sur les sites autrichiens et anglais ont été réalisées par les partenaires. J'ai effectué personnellement les enquêtes sur le site français et l'analyse des résultats de l'ensemble des enquêtes.

1 Introduction

Since the 1990s, multifunctionality has been adopted as a key component of the European Union’s Common Agricultural Policy (CAP) and has increasingly been used in scientific and political debates (Marsden and Sonnino, 2008; Renting *et al.*, 2009). It embraces all goods, products and services created by farming activities (Marsden and Sonnino, 2008), thereby highlighting the non-marketed role of agriculture. More recently, the notion of ecosystem services (commonly defined as the benefits people obtain from ecosystems (MEA, 2005)) appears to have promoted a conceptual shift from multifunctionality of agriculture towards multifunctionality of the agro-ecosystem (Simoncini, 2009), conveying a more biodiversity-oriented perspective of multifunctionality. Moreover, several reports on the 2013 CAP reform have proposed that economic incentives should be introduced to encourage farmers to produce ecosystem services (e.g. European Parliament resolution of 8 July 2010 on the future of the Common Agricultural Policy after 2013 (2009/2236(INI))).

Given multiple available definitions of ecosystem services (see chapter 2, Lamarque *et al*., 2011 for a review) and the need for a precise description to present the concept to stakeholders, we defined ecosystem services as all direct *living* components or processes of natural or managed ecosystem used, consumed or enjoyed (passively or actively) *by humans* before any human *transformation* of ecosystem services. This definition highlights the contribution of the interactions between organisms and the physical environment (Mooney *et al.*, 2009), and also the fact that ecosystem services are the end-products of nature (Boyd and Banzhaf, 2007), and not the results of their human transformation (e.g. forage quality or quantity are the services which provide goods such as milk and cheese). The multiple ecosystem services can be classified following different criteria such as functional (MEA, 2005) or spatial characteristics (Costanza, 2008), decision context (Fisher *et al.*, 2009) or specific context such as the agro-ecosystem (Zhang *et al.*, 2007; Le Roux *et al.*, 2008 – see Figure 1). In an agricultural context this view of multifunctionality includes benefits from ecosystem components and processes for the agro-ecosystem, such as soil fertility, improved water cycling or pest control, as well as benefits from the agro-ecosystem to society (Zhang *et al.*, 2007), rather than focusing solely on agricultural output.

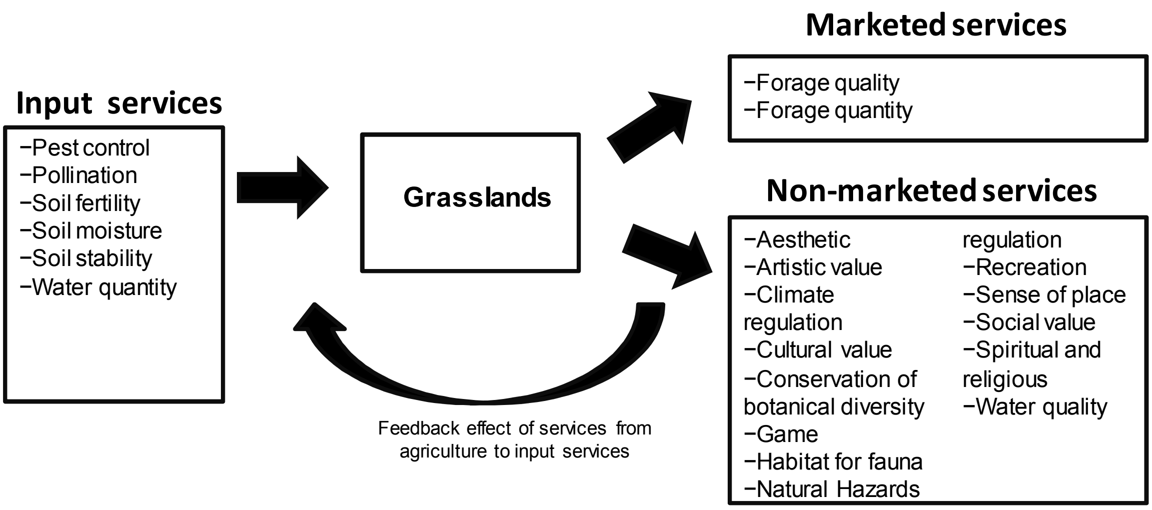


Figure 1: Ecosystem services potentially delivered by semi-natural grasslands – adapted from (Zhang *et al.*, 2007; Le Roux *et al.*, 2008). Input services contribute to biological, physical and chemical processes supporting agriculture, marketed services contribute to agricultural productivity while non-marketed services do not directly contribute to agricultural income (except some specific cases like agro-tourism farms).

The ecosystem properties that underlie ecosystem services depend largely on biodiversity and especially on functional diversity (the presence or abundance of particular functional groups or functional traits) rather than on species number (Hooper *et al.*, 2005; Diaz *et al.*, 2006; Le Roux *et al.*, 2008). In particular, a growing knowledge on plant functional traits (e.g. leaf dry matter content, vegetative height and date of flowering onset) is making it possible to quantify ecosystem services based on responses of functional traits to environmental change and/or effects on ecosystem properties (Diaz *et al.*, 2007; Lavorel *et al.*, 2011). In addition, soil biodiversity and its links with above-ground communities play a significant role in ecosystem services delivery (Barrios, 2007; Turbé *et al.*, 2010). In agro-ecosystems, soil fertility is an important component of soil quality and corresponds to the ability of soils to support plant growth by ensuring the adequate recycling of organic matter, nutrients, soil physical properties and provision of water (Turbé *et al.,* 2010), thereby contributing ecosystem services that support agricultural production. Nevertheless, increasing intensification of agriculture, which is usually associated with increased fertility through fertilizer use and liming, tends to decrease both soil (Bardgett, 2005; Turbé *et al.*, 2010) and above-ground (Walker *et al.*, 2004; Schmitzberger *et al.*, 2005; Klimek *et al.*, 2007) biodiversity. Given such modifications of biodiversity, the supply of ecosystem services is likely to vary with land use and management intensity (Sandhu et al., 2010; Turbé et al., 2010), and it has been proposed that ecosystem services will peak at 'intermediate' levels of intensity (Haines-Young, 2009), as usually found for biodiversity (Bardgett, 2005; Tasser *et al.*, 2005). Finally, sustainable landscape management needs to consider multiple inter-related ecosystem services (Bennett *et al.*, 2009).

As the identification of ecosystem services is motivated by human well-being, stakeholder involvement is particularly important in order to understand people's values and needs (Menzel and Teng, 2009). Moreover, there is a specific need to explore perceptions of grassland ecosystem services in view of current policy change (e.g. CAP reform), which as mentioned above has been gradually shifting its focus from agricultural production to the provision of multiple ecosystem services. However, only few studies of ecosystem services have addressed the identification or perception of ecosystem services by stakeholders (Lewan and Soderqvist, 2002; Pereira *et al.*, 2005; O'Farrell *et al.*, 2007; de Chazal *et al.*, 2008; Quétier *et al.*, 2010). Additional insights knowledge of ecosystem services among stakeholders may also be gained from studies of the perception of biodiversity (Fischer and Young, 2007; Larrère *et al.*, 2007; Buijs *et al.*, 2008), plant uses (Pieroni and Giusti, 2009), and/or the influence of plant diversity on aesthetic appreciation (Lindemann-Matthies *et al.*, 2010). Likewise ethnopedology examines soil and land knowledge by rural communities (e.g. Barrera-Bassols and Zinck, 2003) and studies of traditional or local ecological knowledge identify representations of environmental resources (e.g. Cheveau *et al.*, 2008). However, such studies often focus on a single or few ecosystem services rather than on multiple interlinked services, which remain a significant gap in knowledge.

We propose to address these knowledge gaps by studying perceptions of multiple services by stakeholders, and by placing these perceptions in the broader context of stakeholders' perceptions of the ecosystem through their knowledge of biodiversity and soil fertility. We place special emphasis on soil ecosystem services to address the lack of awareness by stakeholders of their role for the delivery of other ecosystem services (Turbé *et al.*, 2010). Mountain semi-natural grasslands have traditionally delivered multiple ecosystem services in relation to their high levels of above-ground and likely below-ground biodiversity (Figure 1). We used an approach based on interviews with regional experts and local farmers of mountain grasslands to explore: (1) the perception of ecosystem services and the relative importance of different services for different stakeholders of three European mountain semi-natural grassland regions; and (2) in order to build a systemic view, how these perceptions are influenced by stakeholders' knowledge on biodiversity and soil fertility, and by their direct involvement in management.

2 Materials and methods

2.1 Study sites

Permanent grasslands represent a very significant proportion of the European agricultural space (33% of the utilised agricultural area (UAA) in the EU in 2007 – Eurostat, 2010). Species-rich traditionally managed grasslands are strong asset for European society, but they are threatened by changes in land use, intensive management or abandonment (McDonald *et al*, 2000; Gibon, 2005; Spiegelberger *et al*, 2006). In this study, we focused on three European grassland-dominated mountain regions, chosen to represent a gradient in management intensity and associated soil fertility (Figure 2): the French Alps (Villar d'Arène); the Austrian Alps (Stubai Valley); and the English uplands (Yorkshire Dales). These regions are all used primarily for livestock rearing (cattle or sheep) with heterogeneous management intensity and therefore soil fertility within each site, and represent a diversity of agricultural dynamics across European mountain grasslands over the last 50 years.

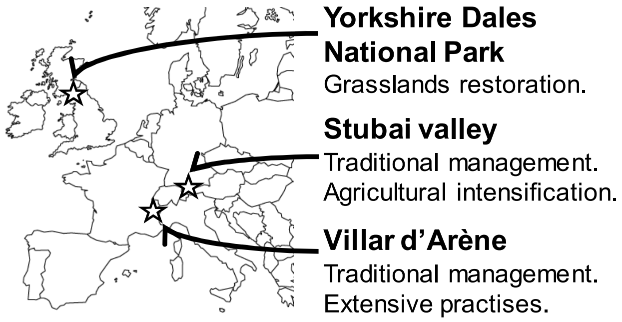


Figure 2: Study areas location and their agricultural intensification characteristics

The upper slopes (above 2500 m) of Villar d'Arène have been extensively grazed for centuries, but the lower slopes have undergone land use change over the last century. Following rural exodus at the beginning of the 20^{th} century, former arable land on terraced slopes (1650-2000 m) was abandoned and transformed into grasslands that are now cut for hay where they are accessible to machinery or grazed. In these grasslands, as well as in those grasslands managed for hay production since the 1700s (1800-2500 m), management practices have remained at low intensity, with low stocking rates, very low manure inputs (every two or three years) and a single annual hay cut. This management mosaic results in distinct patterns of fertility, floristic and functional composition, and associated ecosystem properties (Quétier *et al.*, 2007; Robson *et al.*, 2007). The Stubai Valley was mainly agrarian until the 1970s, but since then the labour force has shifted massively from agriculture to other sectors such as tourism. This has occurred alongside an important structural transition within agriculture from full-time to part-time farming (1970: 57% part-time farmers, 2000: 80% part-time farmers, ISIS, Statistics Austria). Therefore, a dichotomy appears between lightly used high altitude meadows (at and above treeline which lies around 1900 m) where management intensity is determined by accessibility to machinery, and the bottom of the valley where meadows are used intensively, with high rates of fertilizer application and two or three cuts of vegetation per year. Some pastures and meadows are abandoned and colonised by shrubs and trees. The resulting vegetation is a mosaic of forest and diverse grassland types (Tasser *et al.*, 2005; Tasser *et al.*, 2007). Traditionally grasslands in the Yorkshire Dales were used for hay production and livestock grazing, using traditional methods of farming which involved a single annual hay cut and inter-season grazing, with the application of some manure and lime (Smith *et al.,* 2008). However, since the 1950's there has been a shift to more intensive livestock and forage production with high rates of fertilizer application and multiple harvest of grass for silage. In recent years, there have been movements in the area to restore species-rich hay meadows by seeding (input from species rich meadows), controlled cutting and reduced fertilizer use, but silage is still produced. Such variation in management intensity is directly related to plant species richness, functional composition, as well as to soil biological diversity and function (Donnison *et al.,* 2000; Smith *et al.*, 2008; de Deyn *et al.* 2011).

In addition to agriculture, tourism is a dominant economic activity in all three regions, which are recognized for their aesthetic, cultural and conservation value and offer opportunities for recreation. In Austria agro-tourism is well developed, and Villar d'Arène and Yorkshire Dales are parts of national parks. All the differences across the three sites are important to consider in this study as they lead to potentially different supply and demand of ecosystem services across the three regions.

2.2 Stakeholders survey

We aimed to explore ecosystem services identified by different stakeholders related to grassland management and interrelationships between management and ecosystem services. We considered as stakeholders the individuals or sets of individuals who have an interest in ecosystem services because they benefit from them and/or could have an active or passive influence on their delivery (adapted from Billgren and Holmén (2008) and Reed *et al*.(2009)). We aimed to analyse in-depth stakeholders discourses, rather than obtain a representative overview of perceptions and compare them statistically between sub-groups. Therefore, our sample was designed to collect contrasting opinions and points of view (Fischer and Young, 2007; Quetier *et al.*, 2010). The same sampling strategy was used for each study site. Stakeholders were sampled as two groups: (1) regional experts working for governmental institutions, regional institutions or NGOs who represent consumers of their sectors of activity (agriculture, nature conservation, tourism or rural development) and act as decision makers; and (2) local beneficiaries who are consumers (farmers and inhabitants) and/or producers (farmers). Then, within each group we separated stakeholders into two groups, namely those with primary interests in agriculture and those from other socio-economic sectors (tourism, nature conservancy or rural development). All the interviewees were familiar with the regional study site, at least broadly for some regional experts who have expertise in similar agro-ecosytems.

Stakeholders' perceptions of ecosystem services related to management of mountain grasslands were elicited using different methods depending on their origin. Semi-directed individual interviews were used with regional experts because we wanted to elicit mainly factual knowledge, while a group interview was preferred with local beneficiaries because common perception on trends connected to a local context was the focus. Moreover, group interview was the chosen method for farmers to help them speak about unusual issues, and build-up their ideas based on each others responses. Each participant was invited to give his/her opinion on the different themes of the interview guide. Semi-directed interviews are used to collect qualitative data in order to understand the interviewee's point of view. Open-ended questions give medium level of freedom to interviewees to scope their opinions on the subject, but also allow interviewers to reshape questions during the interviews to go into the predefined themes in depth (Grawitz, 2001). Individual interviews and group interviews were considered comparable because in both cases "the emphasis was on questions and responses between the researcher and participants" (Morgan, 1997), a common template was followed (see below), and the group interview did not elicit group interactions (in comparison to a focus group approach). In total, 29 regional expert interviews and three group interviews involving a total of 24 persons were held (Table 1).

Participants selected by reputation or recommendations (snowball strategy) were recruited by phone and invited to an individual interview or a discussion group about the uses and values of grasslands. The term "ecosystem services" was not used in order to prevent participants from trying to collect information before the interview. A common interview guide (Table 2) was used for semi-directive interviews and group interviews across the three regions. Interviews and group interviews lasted between one and two hours and were carried out between summer 2009 and spring 2010. In order to start the discussion and test stakeholders' knowledge and perception of below-ground and above-ground components of grassland ecosystems, the first part of the interview focussed on their descriptions of biodiversity and soil fertility in the context of grasslands of their area. Further questions on relationships with agricultural practices and linkages between the two terms were asked if relevant. The second part of the interview focused on ecosystem services. We decided first to ask to participants to provide a spontaneous list given the previously discussed definition and second to request a ranking for the five most important ecosystem services from a proposed service list discussed with interviewees. This was in order to: (1) check that people understand correctly the concept; (2) analyse stakeholders' perceptions and associations with the term; and (3) potentially complete our ecosystem services list.

		Villar d'Arène	**Stubai valley**	**Yorkshire dales**
Regional experts (Individual interviews)	Agricultural sector	6 *(VAR1)*	6 (include 3 farmers) *(SVR1)*	3 *(YDR1)*
	Non agricultural sector (Nature conservation, tourism,…)	7 (1 tourism, 6 NC *(VAR2)*	3 (2 tourism and 1 NC) *(SVR2)*	4 (NC) *(YDR2)*
Local beneficiaries (Group interview)	Farmers	3 *(VAL1)*	14 *(SVL1)*	4 *(YDL1)*

Table 1: Stakeholders sampling (codes used in the results section refer to the respective individual interviews or group interviews)

Introduction	Can you describe particular characteristics of grasslands?
Soil fertility	a) What is soil fertility? b) How is soil fertility affected by agricultural activities? c) Can agriculture lead to an increase/decrease in soil fertility? d) How could you measure soil fertility?
Biodiversity	a) What is biodiversity? b) How is biodiversity affected by agricultural activities? c) Can agriculture lead to an increase/decrease in biodiversity? d) How could you measure biodiversity?
Relationship	Do you think there is a relationship between soil fertility and biodiversity?
	How do you think farmers/stakeholders have knowledge on soil fertility and biodiversity?
Ecosystem services	a) Do you know the concept of ecosystem services, what does it mean ? (only asked to regional experts) b) According to the definition, could you give me some examples of ecosystem services delivered by mountain grasslands? Any other services? (except local farmers from the Stubai valley) c) Scientists identified some other services, can you comment this list? Could you sort them by order of importance or identify the five most important? d) Are there any links between soil fertility, biodiversity and these services? e) How important is agricultural practice in the supply of ES?

Table 2: Interview guide

2.3 Data analysis

All the interviews were recorded and subsequently analysed. Discussions on biodiversity and soil fertility were analysed using thematic coding. First, broad coding categories were defined (e.g. definition, relation with agricultural management, link with soil fertility) according to our research objectives and questions, and second categories were refined and specified according to the results (e.g. words used to define each notion). All the results were tabulated allowing easy comparison across study regions. Concerning the second section focusing explicitly on ecosystem services, a list of twenty one ecosystem services were pre-selected according to a literature review (Figure 1) and grassland local setting. Then, ecosystem services spontaneously identified or described during the interviews were scored against this list. The five most important ecosystem services ranked by interviewees were analysed without considering their rank order, and aggregated by group of stakeholders and country. This was done to avoid potential errors linked to difficulties met by interviewees in ranking services (Lewan and Soderqvist, 2002).

3 Results

Following our analytical strategy and the interview guide, we analysed successively the understanding of biodiversity and soil fertility by interviewees and their interests and uses of ecosystems (ecosystem services). As no strong differences in biodiversity and soil fertility knowledge across regions or stakeholder groups were observed, we present results overall and specify differences only where relevant.

3.1 Stakeholders' knowledge and understanding of biodiversity

Although the United Nations proclaimed 2010 to be the International Year of Biodiversity, we observed two types of reactions to the question "What does biodiversity mean for you: definition, and critical comment (*"This is a buzzword"* [VAR2]). Four common different criteria appeared in interviewees' definitions: scale, type of organism (plant and animal), species variety and number. All interviewees described biodiversity at the species level, but some also described biodiversity of habitats or landscape, and two regional experts considered multiple scales of biodiversity from genes to landscapes. While stakeholders from Villar d'Arène referred mostly to flora, stakeholders from the Yorkshire Dales and the Stubai Valley spoke more about wildlife in general. One farmer from the Stubai Valley included the diversity of farm animals, but only one regional expert in nature conservation from the Yorkshire Dales specified that biodiversity is both above-ground and below-ground. Interviewees spoke generally about species, but some of them added the adjectives: heritage, rare, common or wild. Terms like number, abundance or richness of species, or the wealth of all living things, were also used. Finally, interactions between organisms were mentioned only by one respondent.

> "*All the different life cycle chains of plants, birds and animals living in the countryside, how they interact together and keeping it as rich as possible.*" [YDR2, group code see Table 1]

Negative impacts of agricultural management on biodiversity were generally recognised, but positive benefits were also identified. Positive management effects on biodiversity were for example: late hays cuts (good for seed dispersal and allowing ground nesting birds to fledge), mowing rather than grazing, reduced and well organized grazing and replacing sheep with cattle which are less selective. "*Mown grasslands are "richer" than grazed grasslands. We can describe it as a decreasing gradient from good to less biodiversity respectively associated to mowing, well organized grazing, badly organized grazing*" [FL2].

Some respondents also discussed increases in common biodiversity rather than rarer species. The role of agricultural management in maintaining open landscapes and landscape diversity was also raised. Negative impacts of agriculture on biodiversity were related to the following practices: intensification of agriculture including heavy grazing, frequent cutting, inorganic fertilizer and slurry application, and pesticide use. But respondents also highlighted that extensification of management and associated low-grazing pressures can reduce biodiversity. Generally, management that is either too intensive or too extensive was considered to be negative. A regional expert in nature conservation also said that the impact of management is not always immediate, so the effect of management practices depends on the time scale of observations [VAR1]. Finally, the difficulty in distinguishing the effects of agriculture from those of abiotic factors such as geomorphology or altitude was noted. Effects of biodiversity on agriculture and why people are interested by biodiversity were also discussed by respondents, but the results are described in the ecosystem services section below.

3.2 Stakeholders' knowledge and understanding of soil fertility

Although the question initially sparked hesitation, low confidence and a need to remember academic definitions, soil fertility was either understood as soil quality or as fertilization effect, and three types of definitions were provided: (1) soil fertility as the ability of soils to sustain plant growth, plant diversity and yield or biomass; (2) the concentration or availability in organic and mineral (N,P,K) elements (given particularly by expert of non-agricultural sectors); and (3) description of activities for maintenance or improvement of soil fertility such as fertilization and liming. An increase in fertilizers, especially livestock manure, was related to improved soil fertility, and some respondents also considered the influence of abiotic factors such as water or moisture, temperature, altitude or solar radiation. *"We can add fertilizers as much as we like, if the soil is dry this will not change anything"* [VAR1]

Only five regional experts mentioned soil microorganisms during the interview, but they did not include this in their definition. When asking farmers from Villar d'Arène about "what is a soil made of" in order to stimulate some responses about soil biota, they said "earth". They explained that in grasslands they are interested in vegetation but they don't work the soil. *"To see if the soil is good or not you need to turn over soil as I do in my vegetable garden. I observed there that the soil is better where it is dry in contrast to a heavy or sticky soil"*[VAL1].

While some respondents differentiate natural fertility from managed fertility, *"Soil fertility is important for highland agriculture where intensive management is impossible"* [SVR1] *"Fertility in the sense… soil for agriculture or soil at natural state?"* [VAL1].

In all cases a relationship between soil fertility and agricultural activity was recognised. All respondents associated decreased soil fertility with reduced biomass or yield, but also decreased feed quality [SVL1]. Agricultural intensification and the uses of fertilizers, manure and lime (in the Stubai Valley and the Yorkshire Dales) were given as examples of how agriculture can increase soil fertility. Conversely, biomass removal by grazing or mowing without fertilization was considered to be a way that agriculture can decrease soil fertility. Only a few experts from Villar d'Arène explained that good agricultural management that is not too extensive and intensive leads to a good balance of soil components. Some experts from the Stubai Valley and a farmer from Villar d'Arène highlighted the fact that intensification of farming is driven by economic constraints. Intensification does not always lead to increased fertilization, but to a change in equipment that promotes soil erosion through compaction and subsequently decreases soil fertility. Finally, methods proposed by interviewees to assess soil fertility were soil analyses and observation of vegetation, i.e. greener vegetation and plants with large leaves such as clover (*Trifolium repens* and *Trifolium pratense)*, rye grass (*Lolium perenne)*, chickweed

(*Stellaria media)*, doc (*Rumex obtusifolius*) and rigwort plantain (*Plantago lanceolata*) were thought to be associated to high fertility. *"One year, a guy writes an « M » with chemical fertilizers in a grassland, and it was visible all summer. Even the difference between land where we put manure and the other can be observed by the difference in grass colour."* [VAL1]

3.3 Stakeholders' knowledge and understanding of ecosystem services

3.3.1 Regional experts

In general the term "ecosystem services" appeared new to respondents, except for two regional experts from Villar d'Arène from nature conservation organisations, two experts from the Stubai Valley working in agricultural sectors, and almost all experts from the Yorkshire Dales, including one who is involved in the UK National Ecosystem Assessment (NEA, 2010). However, the general concept seemed to be understood broadly after a short introduction and definition, although people were more able to identify environmental services which are more linked to human made components of landscapes (e.g. beauty of terraces or small villages) or agricultural activities (see chapter 2, Lamarque et al, 2011 for a definition of the different services concepts) than ecosystem services *sensu stricto* from grasslands. In this section, only ecosystem services coming at least partially from ecosystem components and processes, according to our definition, are presented.

For all regions and stakeholders taken together, 18 of the 21 pre-listed ecosystem services were cited spontaneously (Table 3) after the presentation of our definition. In addition to our list, only air quality was mentioned once by an interviewee [VAR1]. Interestingly, to produce their list of ecosystem services some interviewees used a comparison of grasslands to other ecosystems such as forests or wetlands (*"Water availability is delivered less by grasslands than by wetlands or forests, but it's better than without vegetation soil."*[VAR2]), and selected the service that grasslands deliver more than the other ecosystems. The state of grasslands (abandoned, well managed) or landscape diversity and fragmentation (presence of hedges, trees or a stream) were sometimes discussed as important elements which contribute to ecosystem services such as habitat for fauna or aesthetic value.

Classification	Ecosystem services	Villar d'Arène	Yorkshire Dales	Stubai Valley
Input	Pollination			X
	Soil fertility			
	Soil stability	X		X
	Pest control			
	Soil moisture			
	Water quantity	X	X	X
Marketed	Forage quality	X	X	X
	Forage quantity		X	X
Non-marketed	Conservation of botanical diversity	X	X	X
	Habitat for fauna	X		
	Aesthetic	X	X	X
	Cultural value	X	X	
	Natural hazards regulation	X		X
	Recreation	X	X	X
	Water quality	X	X	X
	Climate regulation/ C-sequestration		X	
	Education			
	Game			
	Sense of place			
	Artistic value			
	Religious and spiritual			

Table 3: Similarities and differences in ecosystem services identified and listed by regional experts of each region. Grey filled cell means mentioned and "X" means mentioned by more than one respondent.

A common set of nine ecosystem services was identified across regions (Table 3) including two out of eight from the input category, one of two marketed services, and six of the eleven non-marketed services. Only regional experts in the Stubai Valley and the Yorkshire Dales identified the three ecosystem services of pollination, forage quantity and climate regulation. Five ecosystem services, namely soil fertility, pest control, game, sense of place and spiritual or religious services, were identified by only one respondent across the three regions. When ranking ecosystem services, as in Lewan and Soderqvist (2002), a discussion arose from some interviewees about the difficulty of doing that due to: (1) the tight interrelationship among some ecosystem services; (2) the extent that some services are more important than other ones; and (3) which standpoint they should take (themselves, society, their institution or organisation).

Classification	Ecosystem services	Villar d'Arène	Yorkshire Dales	Stubai Valley
Input	Pollination	X	X	
	Soil fertility			
	Soil stability	X	X	X
	Pest control			
	Soil moisture			
	Water quantity			X
Marketed	Forage quality	X		X
	Forage quantity	X		X
Non-marketed	Conservation of botanical diversity	X	X	X
	Habitat for fauna	X	X	
	Aesthetic		X	X
	Cultural value		X	
	Natural hazards regulation	X		X
	Recreation	X		X
	Water quality	X	X	X
	Climate regulation/ C-sequestration	X	X	
	Education			
	Game			
	Sense of place	X		
	Artistic value			
	Religious and spiritual			

Table 4: Similarities and differences in ecosystem services considered to be the five more important by regional experts of each region. The lists of ecosystem service were obtained from the combination of the five most important services identified by regional stakeholders in each study regions. Grey filled cell means mentioned and "X" means mentioned by more than one respondent.

Floral diversity, soil stability, water quantity and quality, forage quality, aesthetic value and recreation were all recognised by regional experts of the all three study regions as important ecosystem services to be protected. Some dissimilarities were also observed across regions (Table 3), but the same trends were present between non-market and input services (8 against 2 for the Stubai Valley, 8 against 4 for the Yorkshire Dales, and 8 against 5 for Villar d'Arène interviewees). Interestingly, ecosystem services considered as important by interviewees from the predefined list (Table 3) were not identical to those they mentioned spontaneously (Table 4). Again, when all regions were considered together eighteen services were listed, but two were not common (pest control and sense of place). For example, more regional experts from the Yorkshire Dales considered input services as important, despite the fact that few were listed spontaneously by them. Conversely, non-market services were not considered important, but were frequently associated to the concept of ecosystem services.

3.3.2 Local farmers

Farmers had never heard about the term "ecosystem services" and they did not discuss the definition. In contrast to regional experts, they had difficulties in ranking ecosystem services by importance. Forage quantity and aesthetic value were both ranked as being important by farmers at the three study regions (Table 5). Nine ecosystem services were considered to be important by farmers of only one study site. Only farmers from the Yorkshire Dales considered pollination, soil stability, water quantity, habitat for fauna, sense of place and artistic value as important. Stubai Valley farmers highlighted recreation as being important, and farmers from Villar d'Arène stressed the importance of pest control and soil moisture. Farmers from Villar d'Arène gave preference to input services (3 against 2) and marketed services, while farmers from the Stubai Valley and the Yorkshire Dales considered non-market services to be more important (4 against 6 for the Yorkshire Dales and 0 against 3 for the Stubai Valley). Six ecosystem services from our list, including natural hazards regulation, water quality and climate regulation, did not appear among the five most important ecosystem services of any of the regions.

Classification	Ecosystem services	Villar d'Arène	Yorkshire Dales	Stubai Valley
Input	Pollination		X	
	Soil fertility	X	X	
	Soil stability		X	
	Pest control	X		
	Soil moisture	X		
	Water quantity		X	
Marketed	Forage quality	X	X	
	Forage quantity	X	X	X
Non-marketed	Conservation of botanical diversity	X	X	-----
	Habitat for fauna		X	-----
	Aesthetic	X	X	X
	Cultural value		X	X
	Natural hazards regulation			
	Recreation			X
	Water quality			
	Climate regulation/ C-sequestration			
	Education			
	Game			-----
	Sense of place		X	-----
	Artistic value		X	
	Religious and spiritual			

Table 5: Ecosystem services identified as among the five more important by farmers from each regions. List obtained from the combination of the five most important services identified by local farmers during group interview sessions in each country. In Austria some services (noted "-----") were not proposed during the group interview session. Grey filled cell means mentioned and "X" means mentioned by more than one respondent.

3.4 Interrelationships between biodiversity, soil fertility and ecosystem services

Regional experts as well as local farmers were asked to identify and explain relationships between biodiversity and ecosystem services, as well as between ecosystem services from the list, with a special focus on soil fertility. Because linkages identified by regional experts were similar to those identified by local farmers, results are described overall and differences specified where relevant.

The results on biodiversity and fertility perceptions suggest that interviewees were only moderately aware of relationships between biodiversity and soil fertility. A negative effect of soil fertility on biodiversity was broadly recognised (*"A highly fertile soil will grow grass very well, a poorly fertile soil has the ability to be more diverse in the range of flora that can be found."* [YDR1] "*Generalist species with large leaves and small stems grown on fertile soil*" [VAL1]), but nonlinear relationships were also described ("*There is a link, but it's not simple*", "*generally an increase in fertility means less biodiversity, however it's not always the case"* [YDR1, YDR2 VAR1, VAR2]), such as an "humpback curve" ("*the relationship is positive or negative depending on the level of soil fertility*" [YDR1, YDR2], "*do not manure beyond some limits, because after you change the flora*" [VAL1]). Besides, some regional experts based their explanation on the theoretical humpback curve between species richness and productivity (Grace 1999) that they know from their education. In addition, some respondents argued that factors such as temperature, climate or altitude influence vegetation, and fertility was not always perceived as having a direct effect on biodiversity (*"Each year the forage is different. Dry year plants have more stems and smaller leaves so it's not good for the forage."[VAL1]).* In general, interviewees spoke more easily about plant attributes such as leaf size or colours than about species.

Soil fertility was perceived as having an overall negative relationship with multiple input and non-market services such as soil stability, climate regulation, water quality (due to nutrient leaching), pollination, aesthetic value, cultural services, education and recreation and sense of place. This is notably due to the perceived negative effect of fertilization on biodiversity. Positive links were only perceived with marketed services (forage yield and quality). Only regional experts from Villar d'Arène identified negative links between soil fertility and forage quality, and interviewees from Villar d'Arène and the Stubai Valley considered aesthetic value and forage quality as positively associated, but interviewees from the Yorkshire Dales perceived the relationship as either negative, or positive and negative.

Biodiversity was considered to impact positively on pollination, pest control, aesthetic value and sense of place. Relationships between biodiversity and forage yield were considered to be positive or negative ("*Farming methods which increase biodiversity bring soil fertility down, so methods for biodiversity are bad for productivity" [YDL1]).*

Relationships among ecosystem services were also identified regardless of biodiversity or soil fertility. For example, a decrease in water availability was considered to decrease forage quantity or soil stability, as well as aesthetic value. A regional expert from the Stubai Valley said that "*beautiful flowers are less usable for forage (in term of raw fiber, raw protein and contents)*" [SVR1], and regional experts from Villar d'Arène linked landscape aesthetics to its tidiness and also perceived a relationship with avalanche regulation. Some ecosystem services were also considered by some respondents to have no relationship with other services. For example, flood control was unrelated to any other service.

4 Discussion

We discuss below the results in relation to our two research questions. The first part of the discussion focuses on ecosystem services perceptions, while the second part deals with the perceived links between agricultural management and ecosystem services through fertility level. We finish by examining the implications of our findings for future studies on ecosystem services and provide some recommendations for policy implementation.

4.1 Ecosystem services perceptions: causes and implications

It is well established that ecosystem services are context-dependent (Singh, 2002; Diaz *et al.*, 2006) and that differences in cultural background and agricultural intensification across regions exist. However, this study suggests that perceptions of ecosystem services by regional experts, in terms of identification and ranking, present several commonalities. Nevertheless, difficulties met by interviewees during the ranking exercise in relation to the different standpoints that they could adopt (i.e. adopting a personal view, that of their employers or the presumed point of view of broader society), suggest that people do not have a fixed set of preferences (Lewan and Soderqvist, 2002). In contrast, at the local scale context seems to have a stronger effect on ecosystem service perception because local farmers place importance on different ecosystem services in the different regions. For example, farmers from the Yorkshire Dales, and more strongly those from Villar d'Arène, ranked input and marketed services as being most important, while Stubai farmers placed more importance on non-market services. This result is consistent with the high rate of part-time farmers (80%) in the Stubai Valley, of which a significant number are involved in tourism. Therefore, recreation, cultural and aesthetic values are of high importance to them. At Villar d'Arène, a recent vole (*Arvicola terrestris*) outbreak damaged grasslands and especially mown and fertilized grasslands (as found by Morilhat *et al.*, 2007). Therefore, farmers identified vole control as an important ecosystem service delivered by some undamaged grasslands, whereas pest control was not considered important by farmers of the other regions who were not troubled by voles or other pests. Of note here is that voles, as a component of the ecosystem, are seen as a dis-service (i.e. a negative ecosystem service) because they damage large areas and reduce hay productivity.Differences between ecosystem services considered important by regional experts and local farmers within regions appear to reflect differences in technical (knowledge and background) and local knowledge (generated by practice and observations). This suggests differences in objectives or concerns across stakeholders (e.g., regional experts and local farmers) (Grimble and Wellard, 1997) which could foster divergent priorities among stakeholders for ecosystem management. Such results highlight the need to increase people's awareness of the utility of particular services for sustainable management (Earl *et al.*, 2010).

Some ecosystem services from our list were rarely (by local famer or regional expert from only one region) or never mentioned spontaneously or considered to be important by interviewees. For example, except for soil stability, ecosystem services delivered by soil biodiversity such as soil fertility or soil moisture were rarely identified. This is probably due to the fact that the roles that soils and their biodiversity play in regulating ecosystem processes and the services that they underpin are poorly understood from a scientific perspective (Bardgett, 2005; Dominati *et al.*, 2010; Turbé *et al.*, 2010). These results highlight: (1) the limited ecological understanding and/or awareness by interviewees of some ecosystem services; and (2) the difference between peoples' values and perceived needs (the individual demand) and the services potentially delivered by grasslands (the supply).

Regional experts did not associate some services with the ecosystem service concept, even if they did consider them to be important on the basis of the list they were provided. Services identified spontaneously were more "visible" services, according to Lewan and Soderqvist (2002), such as recreation, aesthetic, natural hazards regulation, while during the ranking exercise "invisible" services such as pollination and soil fertility emerged. This could lead to misunderstandings when these people are exposed to the "ecosystem services" term in the media or policy. If relevant ecosystem services are not defined in detail, it is likely that the concept will be misunderstood by stakeholders, who may therefore not understand the importance of managing those ecosystem services targeted by policy. For example, in the European Parliament resolution of 8 July 2010 on the future of the Common Agricultural Policy after 2013 (2009/2236(INI))), ecosystem services are cited but not defined: "*(…) CAP must place a greater emphasis on sustainability by providing proper economic incentives for farmers to optimise the delivery of eco-system services and further improve the sound environmental resource management of EU farmland (…)*". Therefore, according to readers, incentives may not be attributed for the same ecosystem services (marketed or non-market services could be promoted at the expense of input services). These findings further demonstrate the importance of asking stakeholders to define or explain individually what each service means for them, highlighting that each individual can have a different view of a specific ecosystem service in relation to their uses or interests. For example, water quantity can mean: water availability to irrigate my field, soil water availability for grasses or, freshwater for stock or domestic consumption. In addition, because each person can have different priorities or interests according to the standpoint taken (society, themselves, institution) (Lewan and Soderqvist, 2002), it is important to specify whom interviewee represents during the interview (especially for ranking services) in order to correctly interpret data.

Although interviewees were able to formulate ecosystem services perceptions, it is important to note that the concept itself was not unanimously accepted. On one hand, it prompted debates with several respondents mainly due to an opposition to monetarization (*"Economic value of ecosystem could be dangerous because not all the services are valuable in term of money. Moreover does it mean that if we have money we can destruct nature?"* [VAR2]), and on the utilitarian rather than intrinsic value given to nature (*"Anthropogenic view of ecosystem"*[VAR2]. *"Nature doesn't give service but human use nature. Ecosystem services should be renamed "Interest for nature"* [VAR2] *or "ecosystem exploitation"* [VAR1]). On the other hand, people have different interpretations for the concept. Some interviewees compared it to multifunctionality, positive amenities, externalities, High Nature Value farming, ecological intensification or natural resources. Indeed, it seems that people think more easily in terms of multifunctionality of agriculture than of ecosystem functioning, as suggested by a preferential focus on non-market services rather than on input services. This is probably due to the growing influence of multifunctionality in framing agricultural and rural development policy over the last ten years. In this context, the ecosystem services concept is still emerging. Nevertheless, it is surprising that even interviewees from the Yorkshire Dales followed this pattern since multifunctionality was not well adopted in rural development programmes in the UK (Marsden and Sonnino, 2008). This suggests that the widespread shift from multifunctionality to this new concept is not clear to some regional experts and needs to be better explained. For example, one of the strengths of the ecosystem services approach compared to the agricultural multifunctionality concept is that it can accommodate values outside farming and highlight the dependence of socio-economic activities such as agriculture on the functioning of ecosystems (Simoncini, 2009).

4.2 Systemic perceptions: a way towards sustainable management?

Interviewees expressed rich and diverse perceptions of biodiversity, irrespective of their scientific knowledge (as found by Fischer and Young, 2007). For example, farmers' descriptions of biodiversity are influenced by their animal husbandry activities (*"For us biodiversity is not the colour like you but it's the quality of forage and how cattle take advantage of it".* [FL]) (as found by Larrère *et al.*, 2007). A description based on uses of biodiversity can be interpreted as an ecosystem services approach. This contrasted with their very poor knowledge of soil biodiversity and of soil in general, which is probably because soils and their biodiversity are not visible.

Indeed, soil fertility was often described in terms of fertilization practices and associated vegetation which are the visible elements of soil fertility.

Overall, two kinds of perceptions of linkages between soil fertility, biodiversity and ecosystem services appeared in interviewees' explanations (Figure 3). These were influenced by their knowledge of soil fertility. Either soil fertility was seen as resulting predominantly from fertilization, with effects perceived as being incompatible with biodiversity and with an associated decrease in several ecosystem services (i.e., input and non-market services such as pollination, pest control, aesthetics and sense of place); or soil fertility was seen as a soil property driven by abiotic factors (e.g. altitude and temperature) and agricultural practices which have, within a range of non-extreme values, a positive effect on biodiversity and thereby on multiple ecosystem services. Consistent with Sandhu *et al.* (2010) and Haines-Young (2009), these results suggest that intensification gives more importance to marketed services than to input services which are considered less important because chemical or mechanical inputs substitute ecological processes (bottom of Figure 3). Nevertheless, the link between biodiversity and especially cultural non-market services (e.g. sense of place, conservation of botanical diversity) can be seen either positively or negatively, due to its dependence on personal perception and variation over time (Vira and Adams, 2009). Therefore, it would be interesting to ask interviewees for further details on which aspects of biodiversity (e.g. rare species, species abundance, biodiversity of habitat) influence ecosystem service supply. For example, Lindemann-Matthies *et al.* (2010) found that while people's aesthetic appreciation increased with grassland species richness, this was modulated by the presence of particular species. Interviewees described more often relationships between ecosystem services and biodiversity by speaking about plant functional traits such as "large leaves or dark green grasses" rather than species. This is consistent with scientific results which suggest that functional diversity has overall a greater relevance than species diversity to ecosystem services delivery (Hooper *et al.*, 2005; Diaz *et al.*, 2006; Le Roux *et al.*, 2008).

Finally, while interviewees usually had no problem in perceiving causal relationships between fertility or more generally agricultural management (e.g. mowing), biodiversity and ecosystem services, they did not perceive interrelationships between ecosystem services. Awareness of agricultural effects was not sufficient to frame sustainable management in terms of ecosystem services, although interactions between ecosystem services can strengthen ecosystem resilience and enhance the provision of multiple services (Bennett *et al.*, 2009). Moreover, ignoring interactions could lead to decisions favoring a single ecosystem service, which could decrease biodiversity if the particular service is not directly associated to biological diversity (Vira and Adams, 2009).

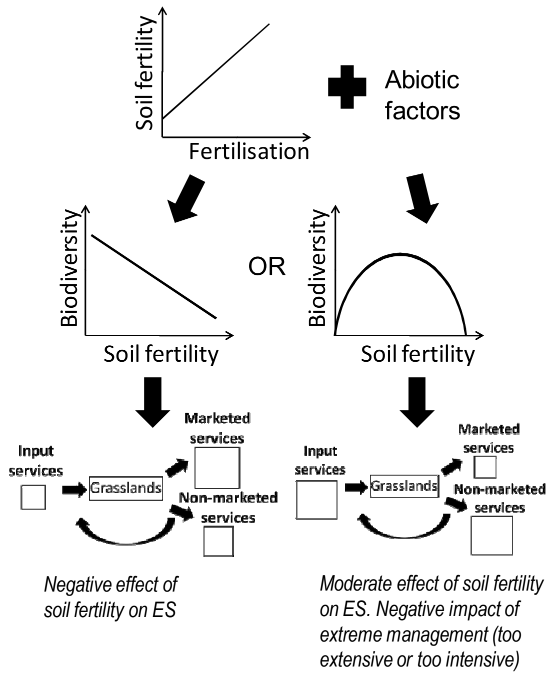


Figure 3: Two kinds of perceptions of the link between soil fertility and ecosystem services (ES), through the relationship between soil fertility and biodiversity, extracted from the analysis of interviewees discourses.

4.3 Research needs and recommendations for policy implementation

While ecosystem services valuation studies are important to identify values involved in decision processes (Brander *et al.*, 2009), they must be complemented by an assessment of stakeholders' perception of the concept (Termorshuizen and Opdam, 2009). Both types of studies are important as they provide complementary information on willingness to trade off conservation of one ecosystem service against another, and awareness and understanding of specific services respectively. Moreover, our results support the need for additional research on demand for and supply of ecosystem services, rather than focusing on supply alone (Termorshuizen and Opdam, 2009). This could help scientists to respond to stakeholders' priorities, but stakeholders points of view are also needed to translate ecosystem functions into ecosystem services. Our results also show the importance of conducting case studies in order to capture local differences in terms of ecosystem service perceptions. In addition, future research should focus more on interrelationships between ecosystem services and systemic representations by stakeholders.

This study showed that it is essential for effective policy implementation and research to have a good understanding of stakeholders' perceptions of ecosystem services, which are themselves linked to their attitudes towards biodiversity management. Our results suggest that achieving sustainable management of grasslands ecosystem services and better acceptance of biodiversity conservation strategies requires: (1) more precise descriptions of which ecosystem services are considered; and (2) improved knowledge of differences in interest and importance of services between stakeholders. We also found that: (3) stakeholder's knowledge of biodiversity and soil fertility influences their perception of agricultural management effects on ecosystem services; and (4) while stakeholders are aware of the effect of agriculture on ecosystem services supply, their knowledge on relationships between ecosystem services are not sufficient and need to be strengthened.

Acknowledgements

We thank all participants of interviews for their contributions and the time they devoted to our study, as well as all the students and people who helped to conduct the interviews. We also thank Philippe Fleury and Fabien Quétier for their useful advice, and the Joseph Fourier Alpine Station for logistical support. This research was conducted on the long term research site "Zone Atelier Alpes", a member of the ILTER-Europe network (ZAA publication n°14) and on the LTER site "Stubai Valley", a member of the Austrian LTSER Platform "Tyrolean Alps". This work was supported by ANR, NERC and FWF through the FP6 BiodivERsA Eranet VITAL project.

References

Bardgett, R.D., 2005. The Biology of Soil : A Community and Ecosystem Approach. Oxford University Press,, Oxford.

Barrera-Bassols, N., Zinck, J.A., 2003. Ethnopedology: a worldwide view on the soil knowledge of local people. Geoderma 111, 171-195.

Barrios, E., 2007. Soil biota, ecosystem services and land productivity. Ecological Economics 64, 269-285.

Bennett, E.M., Peterson, G.D., Gordon, L.J., 2009. Understanding relationships among multiple ecosystem services. Ecology Letters 12, 1394-1404.

Billgren, C., Holmén, H., 2008. Approaching reality: Comparing stakeholder analysis and cultural theory in the context of natural resource management. Land Use Policy 25, 550-562.

Boyd, J., Banzhaf, S., 2007. What are ecosystem services? The need for standardized environmental accounting units. Ecological Economics 63, 616-626.

Brander, L., Gomez-Baggethun, E., Martin-Lopez, B., Verma, M., 2009. Chapter 5: The economics of valuing ecosystem services and biodiversity. TEEB-The Economics of Ecosystems and Biodiversity: The Ecological and Economic Foundations Available at: www.teebweb.org, accessed 26 December 2010.

Buijs, A.E., Fischer, A., Rink, D., Young, J.C., 2008. Looking beyond superficial knowledge gaps: Understanding public representations of biodiversity. International Journal of Biodiversity Science & Management 4, 65 - 80.

Cheveau, M., Imbeau, L., Drapeau, P., Belanger, L., 2008. Current status and future directions of traditional ecological knowledge in forest management: a review. Forestry Chronicle 84, 231-243.

Costanza, R., 2008. Ecosystem services: Multiple classification systems are needed. Biological Conservation 141, 350-352.

de Chazal, J., Quétier, F., Lavorel, S., Van Doorn, A., 2008. Including multiple differing stakeholder values into vulnerability assessments of socio-ecological systems. Global Environmental Change 18, 508-520.

De Deyn, G.B., Shiel, R.S., Ostle, N.J., Mcnamara, N.P., Oakley, S., Young, I., Freeman, C., Fenner, N., Quirk, H. & Bardgett, R.D. 2011. Additional carbon sequestration benefits of grassland diversity restoration. Journal of Applied Ecology, DOI: 10.1111/j.1365-2664.2010.01925.x

Diaz, S., Fargione, J., Stuart Chapin, F., Tilman, D., 2006. Biodiversity Loss Threatens Human Well-Being. PLoS Biology 4, 1300-1305.

Diaz, S., Lavorel, S., de Bello, F., Quétier, F., Grigulis, K., Robson, T.M., 2007. Incorporating plant functional diversity effects in ecosystem service assessments. Proceedings of the National Academy of Science 104, 20684-20689.

Dominati, E., Patterson, M., Mackay, A., 2010. A framework for classifying and quantifying the natural capital and ecosystem services of soils. Ecological Economics 69, 1858-1868.

Donnison, L.M., Griffith, G.S., Hedger, J., Hobbs, P.J., Bardgett, R.D. 2000. Management influences on soil microbial communities and their function in botanically diverse haymeadows of northern England and Wales. Soil Biology and Biochemistry 32, 253-263.

Earl, G., Curtis, A., Allan, C., 2010. Towards a Duty of Care for Biodiversity. Environmental Management 45, 682-696.

Eurostat, 2010. Statistics explained. Agriculture and the environment. (http://epp.eurostat.ec.europa.eu/statistics_explained/index.php/Agriculture_and_the_environment, 22/10/2010)

Fischer, A., Young, J.C., 2007. Understanding mental constructs of biodiversity: Implications for biodiversity management and conservation. Biological Conservation 136, 271-282.

Fisher, B., Turner, R.K., Morling, P., 2009. Defining and classifying ecosystem services for decision making. Ecological Economics 68, 643-653.

Gibon, A., 2005. Managing grassland for production, the environment and the landscape. Challenges at the farm and the landscape level. Livestock Production Science 96, 11-31.

Grace, J.B., 1999. The factors controlling species density in herbaceous plant communities: an assessment. Perspectives in Plant Ecology, Evolution and Systematics 2, 1-28.

Grawitz, M., 2001. Méthodes des sciences sociales. Dalloz.

Grimble, R., Wellard, K., 1997. Stakeholder methodologies in natural resource management: A review of principles, contexts, experiences and opportunities. Agric. Syst. 55, 173-193.

Haines-Young, R., 2009. Land use and biodiversity relationships. Land Use Policy 26, S178-S186.

Hooper, D.U., Chapin, F.S., Ewel, J.J., Hector, A., Inchausti, P., Lavorel, S., Lawton, J.H., Lodge, D.M., Loreau, M., Naeem, S., Schmid, B., Setala, H., Symstad, A.J., Vandermeer, J., Wardle, D.A., 2005. Effects of biodiversity on ecosystem functioning: A consensus of current knowledge. Ecological Monographs 75, 3-35.

Klimek, S., Richter gen. Kemmermann, A., Hofmann, M., Isselstein, J., 2007. Plant species richness and composition in managed grasslands: The relative importance of field management and environmental factors. Biological Conservation 134, 559-570.

Lamarque Pénélope, Quétier Fabien, Lavorel Sandra, 2011. The diversity of the ecosystem services concept and its implications for their assessment and management. Comptes Rendus Biologies. doi:10.1016/j.crvi.2010.11.007

Larrère, R., Fleury, P., Payant, L., 2007. La « nature » des éleveurs : sur les représentations de la biodiversité dans les Alpes du Nord. Ruralia. http://ruralia.revues.org/document1846.html.

Lavorel, S., Grigulis, K., Lamarque, P., Colace, M.-P., Garden, D., Girel, J., Pellet, G., Douzet, R., (2011). Using plant functional traits to understand the landscape distribution of multiple ecosystem services. Journal of Ecology. 99, 135-147. doi: 10.1111/j.1365-2745.2010.01753.x

Le Roux, X., Barbault, R., Baudry, J., Burel, F., Doussan, I., Garnier, E., Herzog, F., Lavorel, S., Lifran, R., Roger-Estrade, J., Sarthou, J.P., Trommetter, M. (Eds.), 2008. Agriculture and biodiversity: benefiting from synergies, Multidisciplinary Scientific Assessment, Synthesis Report. INRA (France).

Lewan, L., Soderqvist, T., 2002. Knowledge and recognition of ecosystem services among the general public in a drainage basin in Scania, Southern Sweden. Ecological Economics 42, 459-467.

Lindemann-Matthies, P., Junge, X., Matthies, D., 2010. The influence of plant diversity on people's perception and aesthetic appreciation of grassland vegetation. Biological Conservation 143, 195-202.

Marsden, T., Sonnino, R., 2008. Rural development and the regional state: Denying multifunctional agriculture in the UK. Journal of Rural Studies 24, 422-431.

MacDonald, D., Crabtree, J.R., Wiesinger, G., Dax, T., Stamou, N., Fleury, P., Gutierrez Lazpita, J., Gibon, A., 2000. Agricultural abandonment in mountain areas of Europe: Environmental consequences and policy response. Journal of Environmental Management 59, 47-69.

MEA, 2005. Millennium Ecosystem Assessment. Ecosystems and Human Well-being: Synthesis. Island Press, Washington DC U.S.A.

Menzel, S., Teng, J., 2009. Ecosystem Services as a Stakeholder-Driven Concept for Conservation Science. Conservation Biology 24, 907-909.

Mooney, H., Larigauderie, A., Cesario, M., Elmquist, T., Hoegh-Guldberg, O., Lavorel, S., Mace, G.M., Palmer, M., Scholes, R., Yahara, T., 2009. Biodiversity, climate change, and ecosystem services. Current Opinion in Environmental Sustainability 1, 46-54.

Morgan D.L., 1997, 2nd Edition. *Focus groups as qualitative research*. London: Sage.

Morilhat, C., Bernard, N., Bournais, C., Meyer, C., Lamboley, C., Giraudoux, P., 2007. Responses of Arvicola terrestris scherman populations to agricultural practices, and to Talpa europaea abundance in eastern France. Agriculture, Ecosystems & Environment 122, 392-398.

NEA. 2010. Website of the UK National Ecosystem Assessment: http://uknea.unep-wcmc.org/, accessed on 22 September 2010

O'Farrell, P.J., Donaldson, J.S., Hoffman, M.T., 2007. The influence of ecosystem goods and services on livestock management practices on the Bokkeveld plateau, South Africa. Agriculture, Ecosystems & Environment 122, 312-324.

Pereira, E., Queiroz, C., Pereira, H.M., Vicente, L., 2005. Ecosystem services and human-well-being: a participatory study in a mountain community in Portugal. Ecology and society 10.

Pieroni, A., Giusti, M., 2009. Alpine ethnobotany in Italy: traditional knowledge of gastronomic and medicinal plants among the Occitans of the upper Varaita valley, Piedmont. Journal of Ethnobiology and Ethnomedicine 5, 32.

Quétier, F., Lavorel, S., Thuillier, W., Davies, I., 2007. Plant-trait-based modelling assessment of ecosystem services sensitivity to land-use change. Ecological Applications 17, 2377-2386.

Quétier, F., Rivoal, F., Marty, P., de Chazal, J., Thuiller, W., Lavorel, S., 2010. Social representations of an alpine grassland landscape and socio-political discourses on rural development. Regional Environmental Change 10, 119-130.

Reed, M.S., Graves, A., Dandy, N., Posthumus, H., Hubacek, K., Morris, J., Prell, C., Quinn, C.H., Stringer, L.C., 2009. Who's in and why? A typology of stakeholder analysis methods for natural resource management. Journal of Environmental Management 90, 1933-1949.

Renting, H., Rossing, W.A.H., Groot, J.C.J., Van der Ploeg, J.D., Laurent, C., Perraud, D., Stobbelaar, D.J., Van Ittersum, M.K., 2009. Exploring multifunctional agriculture. A review of conceptual approaches and prospects for an integrative transitional framework. Journal of Environmental Management 90, S112-S123.

Robson, T.M., Lavorel, S., Clement, J.-C., Roux, X.L., 2007. Neglect of mowing and manuring leads to slower nitrogen cycling in subalpine grasslands. Soil Biology and Biochemistry 39, 930-941.

Sandhu, H.S., Wratten, S.D., Cullen, R., 2010. Organic agriculture and ecosystem services. Environmental Science & Policy 13, 1-7.

Schmitzberger, I., Wrbka, T., Steurer, B., Aschenbrenner, G., Peterseil, J., Zechmeister, H.G., 2005. How farming styles influence biodiversity maintenance in Austrian agricultural landscapes. Agriculture, Ecosystems & Environment 108, 274-290.

Simoncini, R., 2009. Developing an integrated approach to enhance the delivering of environmental goods and services by agro-ecosystems. Regional Environmental Change 9, 153-167.

Singh, S.P., 2002. Balancing the approaches of environmental conservation by considering ecosystem services as well as biodiversity. Current Science 82, 1331-1335.

Smith, R.S., Shiel, R.S., Bardgett, R.D., Millward, D., Corkhill, P., Evans, P., Quirk, H., Hobbs, P., Kometa, S., 2008. Long-term change in vegetation and soil microbial communities during the phased restoration of traditional meadow grassland. Journal of Applied Ecology 45, 670-679.

Spiegelberger, T., Matthies, D., Muller-Scharer, H., Schaffner, U., 2006. Scale-dependent effects of land use on plant species richness of mountain grassland in the European Alps. Ecography 29, 541-548.

Tasser, E., Tappeiner, U., Cernusca, A., 2005. Ecological effects of land use changes in the European Alps. In: Huber, U.M., Bugmann, H.K.M., Reasoner, M. (Eds.), Global Change and Mountain Regions - A State of Knowledge Overview. Springer, Dordrecht, pp. 413-425.

Tasser, E., Walde, J., Tappeiner, U., Teutsch, A., Noggler, W., 2007. Land-use changes and natural reforestation in the Eastern Central Alps. Agriculture, Ecosystems & Environment 118, 115-129.

Termorshuizen, J.W., Opdam, P., 2009. Landscape services as a bridge between landscape ecology and sustainable development Landscape Ecology Volume 24, 1037-1052.

Turbé, A., De Toni, A., Benito, P., Lavelle, P., Lavelle, P., Ruiz, N., Van der Putten, W.H., Labouze, E., Mudgal, S., 2010. Soil biodiversity: functions, threats and tools for policy makers. Bio Intelligence Service, IRD, and NIOO, Report for European Commission (DG Environment).

Vira, B., Adams, W.M., 2009. Ecosystem services and conservation strategy: beware the silver bullet. Conservation Letters 2, 158-162.

Walker, K.J., Stevens, P.A., Stevens, D.P., Mountford, J.O., Manchester, S.J., Pywell, R.F., 2004. The restoration and re-creation of species-rich lowland grassland on land formerly managed for intensive agriculture in the UK. Biological Conservation 119, 1-18.

Zhang, W., Ricketts, T.H., Kremen, C., Carney, K., Swinton, S.M., 2007. Ecosystem services and dis-services to agriculture. Ecological Economics 64, 253-260.

Chapitre 4

Modélisation des services à l'échelle du paysage[9]

Summary

Spatially explicit understanding of the delivery of multiple ecosystem services from global to local scales is currently limited. New studies analysing the simultaneous provision of multiple services at landscape scale should aid the understanding of multiple ecosystem service delivery and trade-offs to support policy, management and land planning.

Here, we propose a new approach for the analysis, mapping and understanding of multiple ecosystem service delivery in landscapes. Spatially explicit single ecosystem service models based on plant traits and abiotic characteristics are combined to identify 'hot' and 'cold' spots of multiple ecosystem service delivery, and the land use and biotic determinants of such distributions. We demonstrate the value of this trait-based approach as compared to a pure land-use approach for a pastoral landscape from the central French Alps, and highlight how it improves understanding of ecological constraints to, and opportunities for, the delivery of multiple services.

Vegetative height and leaf traits such as Leaf Dry Matter Content were response traits strongly influenced by land use and abiotic environment, with follow-on effects on several ecosystem properties, and could therefore be used as functional markers of ecosystem services.

Patterns of association among ecosystem services were related to the dominant traits underlying different ecosystem properties. The functional decoupling between height and leaf traits provided alternative pathways for high agronomic value, as well as determining hot and cold spots of ecosystem services. Traditional land uses such as organic fertilization and mowing or altitude summer grazing were also linked with ecosystem services hot spots, because functional

[9] This chapter is published and can be cited as: Lavorel, S., Grigulis, K., Lamarque, P., Colace, M.-P., Garden, D., Girel, J., Pellet, G., Douzet, R., 2011, Using plant functional traits to understand the landscape distribution of multiple ecosystem services, *Journal of Ecology* **99**(1)**:**135-147.

Remarque : Ma contribution à ce chapitre concerne la réalisation de la cartographie et de l'analyse statistique des résultats de la modélisation spatiale ainsi que la participation à la réflexion méthodologique et l'analyse des résultats.

characteristics supporting fodder production and quality are compatible with species and functional diversity.

Synthesis. Analyses of ecosystem services using plant functional variation across landscapes are a powerful approach to understanding the fundamental ecological mechanisms underlying ecosystem service provision, and trade-offs or synergies among services. Sustainable management of species and functionally diverse grassland could simultaneously aim at conserving biodiversity and locally important ecosystem services by taking advantage of correlations and trade-offs among different plant functional traits.

1 Introduction

Ecosystem services (ES) provide the link between ecosystems - their biodiversity and their functioning - and human society (Millennium Ecosystem Assessment 2005). Most ecosystems provide a diversity of services, such as food and fodder provision, regulation of climate and water quality, pollination, and aesthetic and recreational values (Millennium Ecosystem Assessment 2005). Policy, management and land planning urgently require spatial analyses of multiple ES at global (Naidoo *et al.* 2008), continental (Metzger *et al.* 2006; Kienast *et al.* 2009) and regional (Chan *et al.* 2007; Egoh *et al.* 2009; Eigenbrod *et al.* 2010) scales (Carpenter *et al.* 2009). There is also a critical need for new studies mapping the simultaneous provision of multiple services at landscape scale (Naidoo & Ricketts 2006; Gimona & van der Horst 2007) to advance the understanding of ecosystem service trade-offs and synergies (Bennett *et al.* 2009). Such studies are also required to support the modelling of land change (Verburg *et al.* 2009) and the design of sustainable land architectures (de Groot *et al.* 2009; Turner 2010).

Ecosystem service assessments often make the assumption that ES can be mapped uniquely to land use or land cover (LULC) (Naidoo & Ricketts, 2006; Verburg *et al.* 2009; Eigenbrod *et al.* 2010,), especially at large scales where LULC effects are at best corrected by a few simple modifiers, such as coarse altitude or slope classes, or landscape heterogeneity for which extensive information is available (Kienast *et al.* 2009, Eigenbrod et al. 2010). Yet this approach can introduce errors because it does not account for spatial variability in biophysical variables (e.g. soils, topography) or processes (Eigenbrod *et al.* 2010). For example Eigenbrod *et al.* (2010) demonstrated that mapping ES over England using either land cover or more refined proxies based on strong causal drivers for specific services resulted in a poor fit to primary data, as well as introducing errors in the identification in ES hotspots. While of some use to depict broad-scale patterns of ES delivery in the absence of better data, the use of LULC proxies is also incompatible with the analysis of mechanisms that drive ES delivery because ecosystem functioning often varies across a LULC class due to biophysical heterogeneity (e.g. topography, soil type) or management (e.g. grazing intensity, logging practices) (Bennett *et al.* 2009, Grêt-Regamey *et al.* 2008, Quétier *et al.* 2007, Reyers *et al.* 2009, Willemen *et al.* 2010) and biotic responses to these factors.

We propose to address this limitation by a refined analysis at landscape scale of some of the ecological mechanisms that drive ecosystem service delivery. Ecosystem service delivery has been related to ecosystem biological characteristics (Kremen, 2005), and more specifically to functional traits (Kremen, 2005; De Chazal *et al.* 2008; De Bello *et al.* 2010). In particular, for plants there is growing evidence for the effects of community-level functional traits on ecosystem processes that underlie important ecosystem services (Suding & Goldstein 2008). Following the biomass ratio hypothesis (Grime 1998), community-weighted mean traits, which represent the average trait value for a unit of biomass within a community (Garnier et al. 2004, Violle *et al.* 2007), explain variation in net above-ground primary productivity (specific ANPP, Vile *et al.* 2006; ANPP Mokany *et al.* 2008), litter decomposition under field (Garnier *et al.* 2004) and controlled (Cornelissen *et al.* 1999: Fortunel *et al.* 2009) conditions, digestibility (Pontes Da Silva *et al.* 2007), or soil moisture (Mokany et al. 2008) and water uptake (Gross *et al.* 2008). Effects of functional divergence, i.e. the expected variance in trait values across two random units of biomass within a community (Lepš et al. 2006), have also been hypothesized to operate through functional complementarity (Petchey & Gaston 2006). For example, within-community diversity in plant heights is expected to enhance light capture (Vojtech *et al.* 2008), while diversity in leaf structural and chemical traits would reflect diversity in nutrient acquisition and retention strategies (Gross *et al.* 2007), and therefore affect primary productivity (Schumacher & Roscher 2009) and decomposition (Scherer-Lorenzen 2008). However, their demonstration has so far remained more elusive for such ecosystem processes that appear to be dominated by biomass ratio effects (Díaz et al. 2007; Mokany et al. 2008; but see Schumacher & Roscher 2009). Functional trait data is becoming increasingly available thanks to standardized measurement methods, which have promoted their wide use (Cornelissen et al. 2003), and to large trait data bases (Kleyer et al. 2008, Kattge *et al.* 2010). Quantitative models of ES built from plant traits and environmental variables (Diaz *et al.* 2007) have been used at the ecosystem level to quantify and project ES for current management and future scenarios (Quétier *et al.* 2007). However, such applications have projected ecosystem services using unique values for trait means or divergence and of abiotic factors within a given land use (Quétier *et al.* 2009), ignoring the finer-scale biotic (e.g. plant species composition) and abiotic (e.g. topography and soils) variation within each land use that needs to be considered for a spatially explicit landscape analysis.

Furthermore, existing trait-based analyses have considered ES individually rather than bundles of ES with trade-offs and synergies (Bennett *et al.* 2009), as it is increasingly done in spatial ES assessments (Egoh *et al.* 2009, Eigenbrod *et al.* 2010, Naidoo & Ricketts, 2006, Willemen *et al.* 2010). Eigenbrod *et al.* (2010) highlighted the particularly strong limitations of land use or proxy-based analyses when addressing multiple ES. We believe that using the understanding of relationships between ES and traits should strongly advance the understanding of ES synergies and trade-offs. This would be achieved by using knowledge on associations and trade-offs among traits as captured by plant strategy schemes (Grime 1977, Westoby 1998) and trait spectra analyses (Díaz et al. 2004; Wright et al. 2004; Chave *et al.* 2009).

In this study, we propose a new approach for the analysis of multiple ES delivery in landscapes. We first develop spatially explicit ES models based on plant traits and abiotic characteristics, expanding the trait-based conceptual framework (Diaz *et al.* 2007) (Fig. 1). This framework makes it possible to compare and combine land use, direct (abiotic) and indirect (trait-mediated) effects on ecosystem properties by comparing statistical models incorporating hierarchical combinations of effects. Then 'hot' and 'cold' spots of ES delivery, representing areas of high delivery for multiple vs. low delivery across services respectively, and their determinants in terms of land use and plant traits are analysed combining multiple ecosystem properties. Using interdisciplinary data for a grassland-dominated landscape from the central French Alps, where animal husbandry and tourism are the main activities, we demonstrate how this trait-based approach improves on a pure land use approach, and how it advances understanding of ecological constraints to, and opportunities for, the delivery of multiple services.

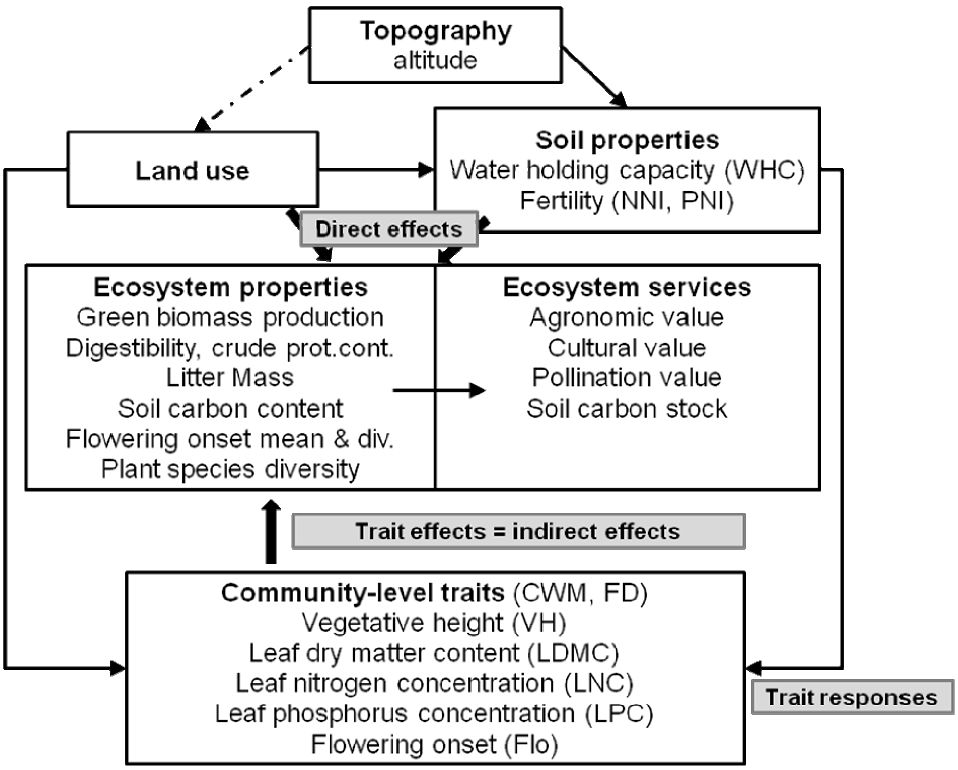


Fig. 1 : Conceptual framework for the analysis of ecosystem properties underlying ecosystem services. The analysis identifies successively the direct effects of land use on ecosystem properties (*LU*); the combined direct effects of land use and abiotic variables on ecosystem properties (*LU+abiotic*); and the combination of

abiotic effects with indirect effects via plant functional diversity (trait community-weighted mean and functional divergence) (*trait+abiotic*).

2 Materials and Methods

2.1 Study site and field measurements

The Lautaret study site (45°03’ N, 6°24’ E) is located in the Central French Alps on the south-facing slopes of Villar d’Arène. The total area is 13 km^2 and the elevation ranges from 1552 to 2442 m a.s.l. A detailed site description can be found in (Quétier, Thébault & Lavorel 2007). Land use legacies can play a key role in determining current vegetation, soil properties and ecosystem functioning (Bruun *et al.* 2001), (Fraterrigo *et al.* 2006), especially in mountain grasslands (Maurer *et al.* 2006). Therefore we considered land use trajectories, the combinations between past and present land use mapped at site level using a combination of cadastral (1810 to present) and aerial photographic data (since 1952) (Fig. 2) (Quétier, Thébault & Lavorel 2007 – see Girel *et al.* 2010 for a detailed analysis of land use history). We analysed eight trajectories, referred to as ‘land use’ henceforth, three on previously cultivated terraces (currently fertilized and mown (LU1), mown (LU2), or unmown and grazed in spring and autumn (LU3)), three on never cultivated permanent grasslands with a multi-century history of mowing (currently mown (LU4), unmown and summer-grazed (LU5), and neither mown nor grazed (LU6) ‘*Festuca* grasslands’ – dominated by the large perennial grass *Festuca paniculata*), one on never mown summer grasslands (> 2000 m) (LU7) and one on steep (> 30°) grazed slopes (LU8). Previous analyses have demonstrated significant differences in soils, plant species and functional composition and ecosystem properties across these land use categories, reflecting both the effects of past land use (presence or absence of cultivation) and current practices (presence or absence of mowing and of fertilization) (Quétier, Thébault & Lavorel 2007; Robson *et al.* 2007). All data were referenced in a Geographic Information System including also a 10-m Digital Elevation Model under ArcGiS 9.2, ESRI.

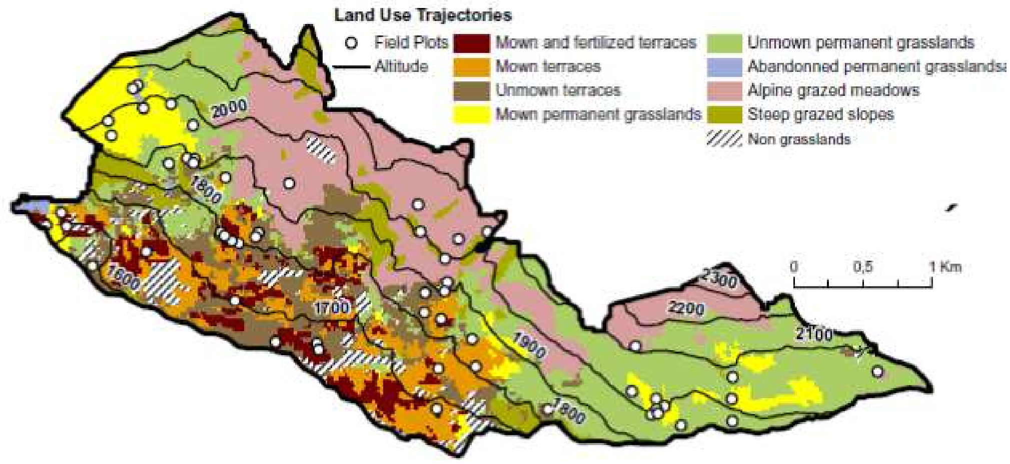


Fig. 2: Field site and land use types. Land use trajectories are the combinations between past and present land use, with three land uses on previously cultivated terraces: fertilized and mown, mown, unmown and grazed in spring and autumn; three on permanent, never-cultivated permanent grasslands: mown, unmown and summer grazed, and neither mown nor grazed '*Festuca* grasslands'; never mown summer grasslands (> 2000 m); steep (> 30°) grazed slopes. Sampled sites are marked with white dots.

Vegetation, plant functional trait, ecosystem and environmental data (Figure 1) were collected for fifty-seven 30 $\times$ 30 m permanent plots stratified by land use (eight categories), landscape sector (four sectors defined based on local toponymy and representing homogenous topography and distance to the village), and altitude within each of these. Vegetation composition surveys used the BOTANAL method to estimate species relative biomass (Lavorel *et al.* 2008). Plant vegetative traits (vegetative height - VH, leaf dry matter content – LDMC, leaf nitrogen and phosphorus concentrations – LNC and LPC) assumed as relevant to ES provision (Quétier *et al.* 2007) were measured for all species making up 80% of cumulated biomass following standard protocols (Garnier *et al.* 2007). For each trait we calculated community-weighted mean (CWM; Garnier et al. 2004) and functional divergence (FD, using the formulation by Mason *et al.* 2003) using the F-Diversity package (Di Rienzo *et al.* 2008). Soil texture, soil total carbon and nitrogen, and nitrogen and phosphorus nutrition indices (NNI and PNI respectively) were also measured in each plot using standard protocols (Garnier *et al.* 2007). Soil water holding capacity (WHC) was calculated using texture and total carbon data (Osty 1971). Radiation was estimated within the GIS using the site's 10-m resolution Digital Elevation model. Green and litter biomass were estimated using calibrated visual estimates (Lavorel *et al.* 2008) in 2007, 2008 and 2009, and analyses used a smoothed mean over the three years with 2008 as a reference. Crude protein content (CPC) of green biomass was estimated using near infrared spectrometry (Pontes Da Silva *et al.* 2007) for a subset of 24 plots distributed across land uses and altitudes. Flowering phenology (date of flowering onset and duration of flowering) was surveyed for all species contributing to 80% cumulated biomass in 2007 and 2008 for a subset of 39 plots distributed across land uses. Date of flowering onset was transformed to growing degree days adjusted to altitude with a 0.6 °C 100 m^{-1} decrease validated with two weather stations located at 1600 m and 2100 m. Functional divergence in flowering dates (FD_Flo) was calculated using a dedicated Excel© Macro (Leps *et al.* 2006).

2.2 Statistical analyses

Variations across the landscape in community-weighted means (CWM) and functional divergence (FD) for the four traits of the vegetative phase were modelled with general linear models (GLM) combining land use (one categorical variable) and abiotic variables (four continuous variables: altitude, radiation, WHC, NNI, PNI). Variation in biogeochemical ecosystem properties (EP) (green biomass production, litter mass, fodder crude protein content, soil C) was modelled using three alternative general linear models (Fig. 1): (i) land use alone (*LU*; categorical variable with eight states), (ii) land use and all abiotic variables (continuous variables) (*LU+abiotic*), and (iii) traits CWM and FD and abiotic variables (continuous variables; *trait+abiotic*) following Diaz *et al.* (2007). The land-use-alone model represents the 'default' model that would be used in the absence of ecological or terrain data, as done in studies using land use as a proxy for ecosystem services (Eigenbrod et al. 2010). The second model combines land use and abiotic effects and provides a purely geographic representation in the absence of ecological knowledge (e.g. Kienast et al. 2009, Grêt-Régamey et al. 2008). The comparison between these first two models identifies effects of abiotic variables that may need to be taken into account in broad-scale ES assessments. Finally, the third model combines trait and abiotic effects as proposed by Díaz *et al.* (2007). The comparison between this model and the land-use-alone model identifies the need for site-based information beyond a land use or land cover proxy, and the comparison with the *land use + abiotic* model assesses the value of additional ecological (trait) information. Given the likely priority of abiotic effects over biotic effects (Grime 1998 – see Díaz *et al.* 2007) a trait-alone model was not considered in the comparison. However, a trait-alone model was also tested in preliminary analyses for those EP (green biomass production and crude protein content) for which significant abiotic effects were retained in the combined trait and abiotic model. It produced very similar results in terms of fit and parsimony to the combined model for green biomass production, whereas for crude protein content the trait-alone model performed considerably worse than the combined model (44 vs. 62% variance explained). Therefore we present only the *trait + abiotic* model, given that for litter and soil carbon content this was actually a trait-alone model (see Table 3).

Simpson species diversity was modelled using the *LU+abiotic* model given that functional diversity should be a consequence of species diversity rather than the reverse (Lepš et al. 2006). Phenological ecosystem properties (CWM onset of flowering and FD_Flo, which in fact are trait functional diversity measures) were modelled using mixed models with land use and abiotic variables as fixed effects (*LU+abiotic* model) and year as a random effect. All analyses were run using Genstat 11th Edition (VSN International) using all subsets regression (abiotic variables, traits, biogeochemistry, species diversity) and residual estimation of maximum likelihood (REML) (phenology) with quality of prediction (adjusted R) and parsimony using the Akaike criterion as criteria for model selection within each model type (*LU+abiotic* or *trait+abiotic*).

2.3 Mapping ecosystem properties and ecosystem services

Abiotic variables (WHC, NNI, PNI), and community-weighted mean (CWM) and functional divergence (FD) for each trait were modelled for each 20 $\times$ 20 m pixel using GLM estimated effects for each land use category and estimated regression coefficients with abiotic variables (step 1). As a second step, ecosystem properties for each pixel were calculated and mapped using model estimates for effects of land use types (*LU* and *LU+abiotic* models), and for regression coefficients on abiotic variables and traits (*LU+abiotic* and *trait+abiotic* models). For each pixel these calculations were applied to mapped estimates of abiotic variables and trait CWM and FD provided by step 1. This second step is critically novel as compared to a direct application of the model by Díaz *et al.* (2007) in that we explicitly modelled the responses of trait community-weighted means and functional divergences to environment prior to evaluating their effects on ecosystem properties. Such an approach is the key to the explicit representation of functional variation across the landscape, as opposed to the use of unique trait values within each land use (see Albert *et al.* 2010).

For each EP we thus produced one map based on pure land use effects (*LU*) and one map based on the combination of abiotic and traits effects (*trait+abiotic*). Given that the number of measured plots was insufficient for splitting into calibration and validation subsets, the two models were compared visually using mapped differences in estimates and comparisons across models of calculated total EP values per land use type.

Ecosystem services were related to ecosystem properties according to indicators identified by stakeholders (Quétier *et al.* 2007, 2010) or experts (e.g. Martin *et al.* 2009) (Table 1). This approach based on social evaluation of ecosystem services rather than on a top-down scientific expert approach (e.g. Millennium Ecosystem Assessment 2005) makes it possible to quantify service provision as perceived by stakeholders (Bryan *et al.* 2010). Although necessarily site-specific (e.g. the negative perception of litter accumulation for cultural value – see Quétier *et al.* 2010 for discussion), such an approach reveals how ecosystems meet local stakeholders' expectations for services. Based on perceptions by stakeholders from the agricultural sector and Martin *et al.* (2009), grassland agronomic value was the sum of green biomass (fodder quantity), fodder quality as indicated by crude protein content, and flowering phenology (mean community onset CWM_Flo and diversity of flowering onset dates FD_Flo, each with a 0.5 weight so as to give an even weight to phenology as compared to fodder quantity and quality). The inclusion of phenology into agricultural value is important because phenology drives management strategies based on the sharp loss of fodder quality once flowering has begun, especially in grasses

(Ansquer *et al.* 2009). Based on perceived indicators (Quétier *et al.* 2010) cultural value was the sum of positive effects of species diversity and flowering diversity (FD_Flo) minus litter mass. A single EP may simply be mapped onto a single ES as for soil carbon content and climate regulation. Overall, following De Chazal *et al.* (2008), we used simple rather than weighted sums of EP to derive ES, because attribution of specific weights would require in-depth analyses of perception and is highly sensitive to both stakeholder sample and context (see also Quétier *et al.* 2009). Also, this method implicitly assumes linear mapping of EP to ES and an exploration of sensitivity of ES projections to their kinds of relationships to EP (Koch *et al.* 2009) is beyond the scope of this study. Ecosystem services maps produced in step 3 were simple sums of maps for relevant EP produced by step 2 (see Table 3) after scaling to a 0-100 baseline and trimming outliers to the 5-95% quantiles (Venables and Ripley, 2002). Given that the entire landscape is used for agriculture production, we chose to keep continuous ES values rather than applying threshold values to assign provision (or not) of an ES to a given pixel (e.g. Chan *et al.* 2007).

2.4 Analysing multiple ecosystem services

The ability of different landscape locations to provide multiple ecosystem services was assessed additively across ES. A given EP could contribute to several ES, e.g. diversity of flowering onset dates (FD_Flo) contributed to agronomic, cultural and pollination services; therefore, to avoid double counts, the multiple ES map was a sum of maps for uncorrelated EP using 0-100 scaled values. To understand trade-offs and synergies underlying the provision of multiple ES, a PCA on sampled plots was used to characterize underlying patterns of correlation among EP. Coordinates on the first two axes of PCA were then calculated for each map pixel using the linear combinations of EP produced by the PCA, and the two corresponding maps represented areas of trade-offs or synergies.

	Ecosystem properties						
Ecosystem Service	*Green biomass*	*Litter mass*	*Crude Protein Content*	*CWM Flowering*	*FD Flowering*	*Species Diversity*	*Soil Carbon*
Agronomic	1		1	½	½		
Cultural		-1		1		1	
Pollination					1	1	
Soil carbon							1

Table 1: Mapping of ecosystem properties to ecosystem services based on stakeholder perception (agronomic value, cultural value; from Quétier *et al.* 2007, 2010) and expert opinion (agronomic value, pollination, soil carbon). The table presents coefficients used for the summing of individual ecosystem properties to a given ecosystem service based on stakeholders' perceptions, given positive (+1) or negative (-1) contributions. The overall positive contribution of phenology to agronomic value was divided into two variables, community mean and functional divergence of flowering dates, with a weight of ½ each (see Methods).

3 Results

3.1 Landscape variations in vegetation functional composition

Community mean traits were strongly driven by land use but also influenced by altitude (Table 2; Appendix S1 and Fig. S1 in Supporting Information). Land use determined community mean vegetative traits directly (Leaf Dry Matter Content, LDMC), indirectly (Leaf Nitrogen Concentration, LNC) or through mixed direct and indirect effects (Vegetative Height VH and Leaf Phosphorus Concentration LPC), with indirect effects resulting from fertility responses. Altitude had additive direct negative effects for LNC and LPC. Mean community onset of flowering responded to land use, with additive delays due to decreased temperatures with altitude. Functional divergence within communities was variable but, with the exception of LPC and onset of flowering, had little relationship to land use or topography (Table 2).

	Variable	Model	R	AIC	d.f.	LUT	altitude	radiation	WHC	NNI	PNI
Abiotic variables	WHC	single				<0.001	0.009	0.024			
		multiple	67.1	62.3	9	<0.001	0.031	-			
	NNI	single				<0.001	0.035	0.015	<0.001		
		multiple	55.8	58.6	9	<0.001	-	-	0.045		
	PNI	single				<0.001	0.003	0.134	0.309		
		multiple	43.5	60.4	9	0.001	0.05	-	-		
	Fertility index	single				<0.001	0.712	0.506	0.06		
		multiple	59.5	56.6	8	<0.001	-	-	-		
Traits	CWM_VH	single				<0.001	0.035	0.024	0.031	<0.001	0.749
		multiple	79.0	67.8	9	<0.001	-	-	-	0.004	-
	CWM_LDMC	single				0.01	0.044	0.261	0.488	0.205	0.011
		multiple	22.4	55.6	8	0.01	-	-	-	-	-
	CWM_LNC	single				<0.001	<0.001	0.272	0.862	0.002	<0.001
		multiple	58.5	62.9	3	-	0.008	-	-	-	-
	CWM_LPC	single				<0.001	<0.001	0.084	0.332	0.177	<0.001
		multiple	80.9	59.6	10	<0.001	0.044	-	-	*0.072*	-
	CWM-Flo	single				0.012	0.259	0.201	0.072	0.038	0.232
		multiple	45.2	21.5	6	0.012	-	-	-	-	-
	FD_VH	single				0.007	0.023	0.417	0.713	0.369	0.024
		multiple	24.2	64.9	8	0.007	-	-	-	-	-
	FD_LDMC	single				0.268	0.627	0.279	0.415	0.198	0.475
		multiple	NA	NA	NA	NA	NA	NA	NA	NA	NA
	FD_LNC	single				0.072	0.465	0.2	0.64	0.021	0.233
		multiple	12.7	65.8	8	*0.072*	-	-	-	-	-
	FD_LPC	single				<0.001	0.018	0.263	0.517	0.173	<0.001
		multiple	49.9	59.9	10	<0.001	0.004	-	0.015	-	-
	FD_Flo	single				0.937	0.001	0.957	0.271	0.24	0.161
		multiple	35.5	31.2	1	-	0.001	-	-	-	-

Table 2 – Summary of statistics from General Linear Models of abiotic variables and functional diversity components, trait community-weighted mean (CWM) and functional divergence (FD). For onset of flowering we used Residual Estimates Maximum-Likelihood models with year as a random effect. Single-term estimates and combined model selection were obtained using 'all subsets regression' with adjusted-R and Akaike criterion (AIC) as model ranking criteria. %var: percentage variance explained, d.f.: number of degrees of freedom. LUT: land-use type, WHC: water holding capacity, NNI: nitrogen nutrition index, PNI: phosphorus nutrition index, GFI: generalized fertility index, VH: vegetative height, LDCM: leaf dry matter content, MNC: leaf nitrogen concentration, LPC: leaf phosphorus concentration, Flo: date of onset of flowering, NA: model not available. Preliminary variable selection was used to choose the best descriptors of fertility for each response variable. Grey cells indicate variables that were not relevant for particular analyses. p-values are indicated for each response–explanatory variable combination. – indicates variables that were not retained in best multiple regression models.

	Variable	Model	R	AIC	d.f.	LUT	altitude	radiation	WHC	NNI	PNI	CWM_VH	CWM_LDMC	CWM_LNC	CWM_LPC	FD_VH	FD_LDMC	FD_LNC	FD_LPC
Ecosystem properties	Green biomass	single				<0.001	0.562	0.043	0.001	<0.001	0.024	<0.001	0.016	<0.001	<0.001	0.06	0.1	0.037	0.036
		LU	63.4	73.6	8	< 0.001													
		LU+ABIOTIC	74.2	63.0	10	< 0.001	-	-	0.029	0.039	-								
		TRAIT+ABIOTIC	69.9	58.6	4		-	-	0.004	-	-	< 0.001	-	< 0.001	-	-	-	-	-
	Litter mass	single				<0.001	0.002	0.09	0.023	0.091	0.036	<0.001	0.507	0.074	<0.001	<0.001	0.157	0.163	0.009
		LU	66.0	57.4	8	< 0.001													
		LU+ABIOTIC	NA	NA	NA	NA	NA	NA	NA	NA	NA								
		TRAIT+ABIOTIC	61.2	56.4	6		-	-	-	-	-	< 0.001	0.02	-	< 0.001	0.03	-	-	0.03
	Crude Protein Content	single				0.016	0.631	0.96	0.048	0.327	0.216	0.047	0.682	0.836	0.074	0.573	0.433	0.544	0.339
		LU	42.7	23.9	7	0.016													
		LU+ABIOTIC	NA	NA	NA	NA	NA	NA	NA	NA	NA								
		TRAIT+ABIOTIC	62.4	26.1	4		-	-	0.002	-	-	< 0.001	0.008	-	-	-	-	-	-
	Soil carbon	single				0.199	0.052	0.307	0.463	0.362	0.077	0.679	0.4	0.02	0.004	0.053	0.985	0.09	0.051
		LU	9.1	65.1	8	*0.199*													
		LU+ABIOTIC	23.4	58.3	10	0.046	0.009	-	0.007	-	-								
		TRAIT+ABIOTIC	30.7	59.0	3		-	-	-	-	-	-	< 0.001	-	< 0.001	-	-	-	-
	Species diversity	single				0.001	0.137	0.201	0.16	0.004	0.91								
		LU+abiotic	31		10	-	-	-	-	0.004	-								

Table 3 - Summary of statistics from General Linear Models of ecosystem properties from abiotic variables and functional diversity components, trait community weighted mean (CWM) and functional divergence (FD). *LU*: simple land use model, *LU+abiotic*: land use and abiotic variables model, *trait+abiotic*: trait and abotic variables model. Single-term estimates and combined model selection were obtained using 'all subsets regression' with R^2 and Akaike criterion (AIC) as model ranking criteria. %var: percentage variance explained, d.f.: number of degrees of freedom. LUT: land-use type, WHC: water holding capacity, NNI: nitrogen nutrition index, PNI: phosphorus nutrition index, VH: vegetative height, LDMC: leaf dry matter content, LNC: leaf nitrogen concentration, LPC: leaf phosphorus concentration, NA: model not available. As GFI is a linear combination of NNI and PNI, their inclusion into models is exclusive. Preliminary variable selection was used to choose the best descriptors of fertility for each response variable. Grey cells indicate variables that were not relevant for particular analyses. p-values are indicated for each response–explanatory variable combination. – indicates variables that were not retained in best multiple regression models.

3.2 Direct and indirect effects of land use and abiotic factors on ecosystem properties

Models including abiotic factors or traits provided overall better predictions of EP than land-use-alone models, with greater nuances on the predicted effects of land use changes such as cessation of fertilization or mowing (Table 3; SI App., Fig. S2). The *trait+abiotic* model was also the most parsimonious overall for green biomass and soil carbon, while both the *trait+abiotic* and *LU* models had similar empirical support (i.e. differences in AIC<2) for litter mass, and the *LU* model was most parsimonious for crude protein content in spite of a very large increase in prediction ability (adjusted-R increasing from 43 to 62 from the *LU* to the *trait+abiotic* model).

Green biomass production, predicted by mean community traits VH and LNC and soil water holding capacity (WHC), was highest in fertilized and mown terraces and in unmown Festuca grasslands, and least in unfertilized terraces and summer grasslands (Fig. 3a). Production was reduced by cessation of fertilization or of mowing in terraces that both promoted shorter and nitrogen-poorer plants, but it increased by cessation of mowing in old grasslands due to the dominance by the large grass *Festuca paniculata*. Fodder quality, predicted by mean community traits VH and LDMC and WHC, was significantly reduced by cessation of mowing, which promoted plants with denser tissues (higher LDMC), both in terraces and in Festuca grasslands, and improved by fertilization, which increased plant stature and decreased leaf density (LDMC) in terraces (Fig. 3b). Litter mass, predicted by VH, LPC (with CWM and FD for both) and LDMC (CWM only), was greatest following cessation of mowing in both terraces and old grasslands (Fig. 3c). For terraces especially, as well as for other grazed grasslands, litter significantly decreased with altitude, reflecting a decrease in CWM_LPC. Soil carbon stocks, predicted by mean community traits LDMC and LPC, were greatest in mown grasslands, especially in fertilized ones, and in summer grasslands (Fig. 3g). They decreased with altitude following CWM_LPC, especially in mown *Festuca* grasslands and summer grasslands, which were also those grasslands with lower production. Plant species diversity increased with soil nitrogen availability (NNI), which reflected mainly land use and a small effect of altitude through effects of WHC on NNI (Fig. 3f; Table 3).

3.3 Landscape patterns in ecosystem service provision

Ecosystem services patterns were comparable between the pure land-use and the trait-based models, although as for EP, absolute effects of land use changes were moderated by trait-based models (Appendix S1). Agronomic value was highest for summer grasslands, which combined high fodder quality and diverse flowering phenology, but had low production due to short vegetation stature (Fig. 3i). Fertilized and mown terraces also had high agronomic value by combining high fodder quantity (green biomass) and quality resulting from tall stature and high LNC, but less diverse flowering dates. *Festuca* grasslands, especially when unmown, had a lower value in spite of their high stature and production, due to their poor fodder quality resulting from low LNC. Unmown terraces and steep slopes had the poorest value with low scores for all four EP. Overall, agronomic value increased with altitude, which had positive effects on all four EP, especially flowering mean date and diversity. Cultural value was high for mown grasslands, especially in fertilized, and mown terraces, and for summer grasslands, which combined high species diversity, highly diverse flowering phenology and low litter mass,; it was lowest for unmown grasslands, especially *Festuca* grasslands, with the opposite attributes (Fig. 3j). This was a direct land use effect for species diversity but a trait-based effect through litter accumulation associated with high LDMC and tall vegetation in unmown *Festuca* grasslands (Table 3). Climate regulation through soil C sequestration was approximated by soil carbon stocks (soil C) as described above. Pollination followed a pattern close to that of cultural value, as species diversity and diversity in flowering dates were common to these services, while species diversity was strongly negatively correlated with litter (R^2 = 0.98, $p < 0.001$). Total regulation value, combining soil C stocks and pollination, was highest in mown (inter alia) and summer grasslands, with maximum values for fertilized terraces (due to high C stocks and species diversity) and summer grasslands (with high values for all EP) (Fig. 3i). It was lowest for unmown terraces, followed by unmown *Festuca* grasslands, both having low pollination value resulting from low species diversity and, for unmown terraces, particularly low C stocks due to low LNC.

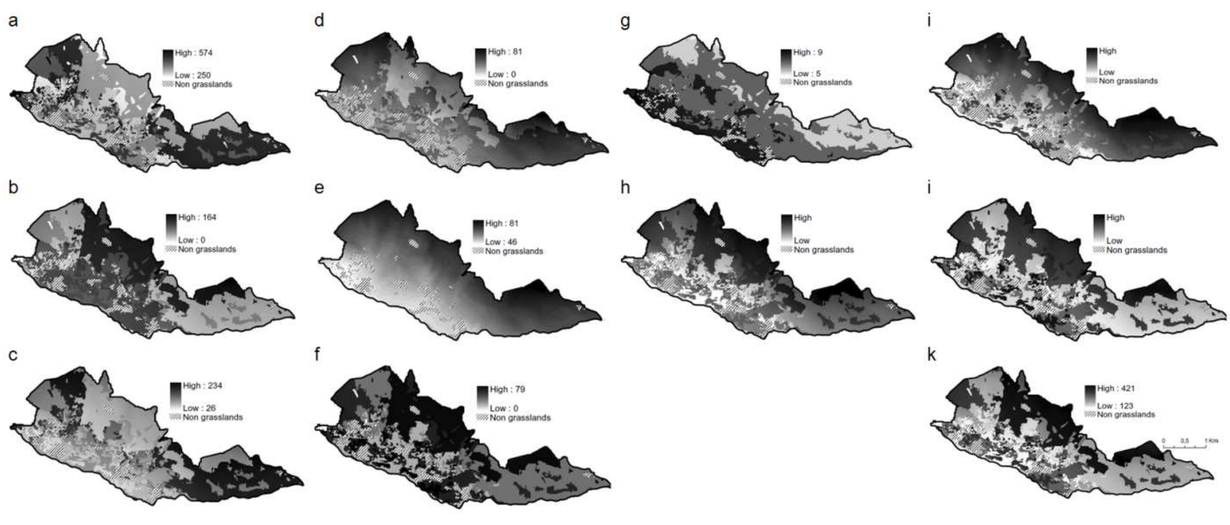


Fig. 3: Modelled distributions of ecosystem properties and ecosystem services. Ecosystem properties: (a) green biomass production (g m^{-2}), (b) fodder crude protein content (g kg^{-1}), (c) litter mass (g m^{-2}), (d) mean date of flowering onset (Julian day), (e) functional divergence of date of flowering onset (unitless), (f) species richness, (g) soil carbon concentration (%). Ecosystem services: (h) pollination value = (e) + (f), (i) agronomic value = (a)+ (b) + ½(d) + ½(e), (j) cultural value = (e) + (f) – (c), (k): total ecosystem service value = (a) + (b) + (c) + (d) + (e) + (f).

3.4 Provision of multiple ecosystem services

The models summing EP showed that fertilized and mown terraces offered the greatest provision and synergy among ecosystem services (Fig. 3k). Summer grasslands were also ES hot spots, despite their low production, which decreased their agronomic value. Mown but unfertilized terraces and mown permanent grasslands showed similar patterns, but with lower provision intensity for all services. In contrast, unmown *Festuca* grasslands were areas of trade-offs among services, with large production potential but low cultural and soil C stocks value. Steep slopes were also ES trade-off areas with lower agronomic value and low C stocks, but higher cultural and pollination values. Finally, unmown terraces delivered the least services, with low provision of all ES. Overall, multi-service patterns were strongly consistent between the pure land-use and the trait-based models (Appendix S1, Fig. S3g-h), although the trait-based model highlighted increased ES with altitude within land use types. The PCA of EPs elucidated these synergies and trade-offs (Fig 4, Fig.S4). The first axis was driven by contrasts between on the one hand high plant diversity, fodder quality (CPC) and soil C in terraces and summer grasslands, and on the other hand high litter accumulation and low diversity of flowering phenology in unmown *Festuca* grasslands. This axis therefore represented contrasts in cultural value, but also potential conflicts among components of regulation services soil C stocks and pollination. The second axis was mainly driven by contrasts in green biomass production from fertilized terraces (highest production) to summer grasslands (lowest production). It also highlighted a trade-off among fodder quantity and quality contrasting high production of poor quality in unmown *Festuca* grasslands with low production of better quality in summer grasslands, although fertilized terraces had high values for both, and thereby also high agronomic value. Finally, orthogonality between cultural value (axis 1) and production (axis 2) indicated the possibility of reconciling both objectives.

Fig. 4: Projected maps of contrasts among ecosystem properties following ordination by PCA (1) Left panel: axis 1 predominantly associated with cultural value, (2) right panel: axis 2 predominantly associated with green biomass production

4 Discussion

4.1 Indirect land use effects on ecosystem services through plant functional traits

Land use or land cover are a practical but imperfect surrogate for ecosystem service assessment (Eigenbrod *et al.* 2010). This is the first study identifying direct and indirect effects of land use and associated abiotic environmental variables on ES using alternative models for ecosystem properties at landscape scale (Fig. 1). All modelled EP showed a direct land use signal. Adding abiotic variables describing topography (altitude) and soil quality (fertility and water holding capacity, themselves related to land use) (*LU+abiotic* model), or representing indirect effects through plant functional traits (*trait+abiotic* model) improved models by often similar levels. With the exception of soil carbon, which was also poorly modelled by land use alone (*LU* model), all EP were remarkably well explained by the statistical models, especially trait-based (*trait+abiotic*) or full abiotic (*LU+abiotic*) models, which afforded better prediction and, in all but the case of litter, equal or better parsimony than the pure land use model (*LU*). Overall, the best *trait+abiotic* models afforded prediction of 60 to 70% of the variance in EP, with usually two traits and often soil properties (WHC in most cases, nitrogen fertility for the *LU+abiotic* model of green biomass production) and altitude (in the *LU+abiotic* model).

Such a continuous quantification of land use effects within a single land cover type (permanent grasslands) goes one step further than categorical modifiers based on land condition (Naidoo & Ricketts 2006; Reyers *et al.* 2009); but see (Grêt-Regamey *et al.* 2008; Willemen *et al.* 2010). Detailed models including abiotic and/or trait effects captured abiotic heterogeneity within land use types, e.g. a 25% variation in green biomass production, litter accumulation or soil C. Green biomass production measurements for an additional set of 34 independent points in 2010, covering a slightly greater altitudinal range for summer grasslands (100 m higher), validated the representativity of our core sample and predictions by the *trait+abiotic* model (significant regression between observed and predicted green biomass, $p = 0.005$). Detailed models also showed that simple land use models overestimated management change impacts by neglecting increases in predicted EP with altitude within land use types, with marked effects especially for summer grasslands and steep slopes. Altitude effects were detected for all EP either directly in abiotic models (*LU+abiotic*) or indirectly in trait-based models (*trait+abiotic*) through the influence of altitude on community traits (CWM) and field capacity (WHC), and were additive to land use, which is also determined by topography at this site as in other mountain systems (Mottet *et al.* 2006; Gellrich and Zimmermann, 2007). These results confirm that for ES assessments in mountains topography, and especially altitude and its effects on bioclimate, must be taken into account in addition to land cover (Grêt-Regamey *et al.* 2008; Kienast et al. 2009). Moreover, the prominent role of WHC in our models emphasizes important effects of current, and especially past, land use on soils. These include fine soil loss and increased stoniness resulting from past cropping on terraces (Bakker *et al.* 2008), long-term effects of organic fertilization on terraces (Robson *et al.* 2007), as well as continued export of organic matter through mowing, which has over the course of history concerned the entire landscape except summer grasslands and steep slopes.

Trait-based models are data-intensive, especially when considering intraspecific trait variation in relation to land use (Garnier *et al.* 2007), but data collection over entire landscapes can be facilitated by standardized and rapid methods (Cornelissen *et al.* 2003; Lavorel *et al.* 2008). For applications such as mapping of ecosystem properties and ecosystem services, trait measurements for randomly sampled individuals (Gaucherand & Lavorel, 2007; Baraloto *et al.* 2009) or for entire swards or canopies (Stewart *et al.* 2001) offer an interesting alternative to the tedious collection of species-level trait data. Landscape- and especially regional-scale applications can also now strongly benefit from the availability of plant functional trait data bases (Kleyer *et al.* 2008, Kattge *et al.* 2010), although caution is warranted with respect to trait variability in response to especially fertility (Lavorel *et al.* 2009). Such data bases will make it possible to assess ecosystem service provision at regional scale by coupling trait and vegetation data bases. Finally, remotely sensed trait surrogates such as spectral signatures of leaf chemistry (Ustin & Gamon 2010) also offer great promise for the application of such trait-based models over large scales.

4.2 Ecological mechanisms underlying ecosystem service responses to land use

Trait-based assessments of global change effects on ecosystems and ecosystem services can reduce uncertainty in projections of land futures (Díaz *et al.* 2007). Prediction of ES change through traits hinges on overlaps of response and effect traits, where traits that determine response to abiotic and land use changes are equal or correlated to traits determining effects on ecosystem functioning (Lavorel & Garnier 2002). Here all vegetative traits responded strongly to land use, except LNC, which had an indirect response through fertility effects. These same traits underpinned relevant EP, thereby providing a link from land use to EP. There is increasing evidence for such overlaps in response and effect traits (Suding & Goldstein 2008), of which this is the first landscape-scale demonstration. In addition, we were able to integrate abiotic (topography and soils) and land use effects with a parsimonious set of traits, namely plant height and key leaf traits associated with plant resource economy (Díaz *et al.* 2004). These traits have demonstrated links to biomass production, litter decomposition, fodder quality or soil water retention from species (Kazakou *et al.* 2006; Pontes Da Silva *et al.* 2007) to community level (Garnier *et al.* 2004; Garnier *et al.* 2007; Gross *et al.* 2008; Fortunel *et al.* 2009). Vegetative height and LDMC were strong response traits with effects on several EP, and could therefore be used as functional markers of ES change (Garnier *et al.* 2004). Considering landscape distribution of EP in response to land use and abiotic factors requires working at community level, where trait responses and effects are indicated by community-weighted means and functional diversity (Diaz *et al.* 2007; Garnier *et al.* 2007). Analyses of this landscape-wide data set confirmed the greater relevance of CWM traits than of functional divergence identified for a subset of 15 plots with similar altitudes (Diaz *et al.* 2007). Only for litter accumulation did the inclusion of FDs for vegetative height and leaf phosphorus concentration markedly improve the prediction from models using CWM traits (61 vs. 44% variance explained). Negative effects of FD on litter accumulation suggested improved decomposition of more diverse mixes of litter types (Gartner & Cardon 2004; Scherer-Lorenzen 2008).

4.3 Assessing multiple ecosystem services

Ecosystem services were even-weight sums of relevant EP, and likewise for the assessment of multiple services (De Chazal *et al.* 2008). Alternative methods may use weights elicited from stakeholders (Gimona and van der Horst, 2007) (for example, farmers at this site rank fodder quantity, phenology and quality differently depending on field functions in their farming system) or different weights across stakeholder groups (see (De Chazal *et al.* 2008), or across alternative future scenarios (Quétier *et al.* 2009). The following discussion focuses on the benefits of plant functional trait information to understand the mechanisms underlying ES provision. Through its component EP, agronomic value was influenced evenly by vegetative height and leaf traits (LNC and LDMC being negatively correlated), with soil WHC and altitude as modifiers. These traits, as well as WHC, propagated a strong land use signal and a fairly strong altitude signal (Table 3). The negative correlation between green biomass production (fodder quantity) and Crude Protein Content (fodder quality) (PCA axis 2) reflects opposite effects of plant height on these two EP and captures effects of *Festuca paniculata* and other tall grasses with poor nutritive quality, especially after flowering, in contrast with smaller species of high value such as legumes (e.g. *Astragalus danicus*, *Oxytropus campestris*) and some dicots (e.g. *Helianthemum grandiflorum*, *Potentilla aurea*) found in summer grasslands. Having species with tall stature and/or high LNC (e.g. *Dactylis glomerata*, *Heracleum sphondylum*, *Onobrychis montana*), fertilized terraces scored high for both quantity and quality. Height and leaf traits such as LNC have indeed been shown to be independent axes of functional variation over continents (Diaz *et al.* 2004) and for this site (Gross, Suding & Lavorel 2007). Diversity of flowering dates (PCA axis 1) added a dimension of variation in agronomic value by being independent from these vegetative traits. Such a combination of independent EP based on independent traits supported the overall value of summer grasslands in spite of their low production, or of fertilized terraces in spite of less diverse flowering dates. Cultural and regulation values shared similar patterns through common EP species diversity and flowering diversity, and the negative correlation ($R^2 = 0.89$, $p < 0.001$) between litter (negative component of cultural value) and soil C (positive component of regulation value). Cultural value was strongly influenced by the well-known negative correlation between litter and species diversity (PCA axis 1; $R^2 = 0.97$, $p < 0.001$), with an additional positive altitude effect through flowering diversity. High cultural value could be attained alternatively with short height (summer grasslands) or with high LPC (fertilized terraces). The regulation value was influenced by two leaf traits LDMC and LNC, with an additional positive altitude effect through flowering diversity. The negative correlation among these leaf traits afforded alternative pathways to increased soil C in fertilized terraces (high LNC), in lower unfertilized terraces and unmown *Festuca* grasslands (high LDMC), and in the lower part of summer grasslands (higher LDMC and LNC). Lower unfertilized and mown terraces and unmown *Festuca* grasslands had higher regulation than cultural value due to this higher soil C.

Consistent with other recent studies, there was a landscape-scale diversity of associations among different types of ES (Chan *et al.* 2007, Egoh *et al.* 2009, Naidoo *et al.* 2008). Service hotspots, with synergy among nearly all services, were fertilized terraces and summer grasslands, which currently represent 5 and 23% of the landscape, respectively. Conversely, unmown *Festuca* grasslands, which represent 28% of the landscape, appeared as areas of trade-offs among services. Unmown terraces (11% of the total area) were services cold spots with low provision for all services, yet our analysis did not consider their agronomic function in terms of spatial complementarity during the annual cycle (Andrieu *et al.* 2007). Ecosystem services hot spots coincided with higher species and functional diversity (Fig. 3, Fig. S1), while areas of ES trade-offs and cold spots were least diverse, suggesting that, unlike in other regions and especially with more intensive agriculture (Chan *et al.* 2007), sustainable management could simultaneously conserve biodiversity and locally important ecosystem services. The synergy among multiple ES was facilitated by both the independence of components of agronomic (green biomass production) vs. cultural and regulation services (litter and species diversity) (orthogonal ordination in the PCA), and the common and/or positively correlated EP contributing to cultural and regulation services (plant diversity, soil C), providing the mechanisms for how at multifunctionality hot spots different ES enhance one another (Bennett *et al.* 2009; Willemen *et al.* 2010). These patterns of independence or conversely correlation were in part related to dominant traits underlying each service. Vegetative height, which determined green biomass production and fodder quality, was a key driver of agronomic value whereas leaf traits played a stronger role for components of regulation and cultural values (soil C, litter). The functional decoupling between these two sets of traits thus contributes not only to agronomic value but also to high multiple ES delivery by fertilized and mown terraces – and conversely to the low score for unmown terraces, with the other land use types scoring high for one but not another service. Consequently, production can be enhanced by moderate organic fertilization without degrading other ecosystem services and the biodiversity that underlies them, as long as appropriate leaf traits are promoted. The future vulnerability of ecosystem services hotspots will also be directly linked to land use and possible climate change effects on plant traits (Quétier *et al.* 2007).

Conclusion

Models of ecosystem services using abiotic variables and plant traits rather than land use alone afford refined representation of relevant ecosystem properties. They also unravel mechanisms controlling ecosystem service delivery, and trade-offs or synergies in provision of multiple ES. Trait-based approaches may be generalized to services provided by other organisms than plants (e.g. pollination, pest control) (De Bello *et al.* 2010). Alternative methods to simple statistical models include structural equation models (Grace, 2006) and process models (Nelson *et al.* 2009) and more complex approaches could be considered for aggregation of ecosystem properties and of ecosystem services to address multifunctionality. In a subalpine grassland landscape traditional land uses such as organic fertilization and mowing or altitude summer grazing supported ecosystem services hot spots because functional characteristics supporting production and fodder quality are compatible with species and functional diversity. Conversely, key vulnerabilities are expected from land change that decreases biodiversity and promotes plant types associated with ecosystem services cold spots and/or strong trade-offs among services. The relevance of this model to broader and more diverse landscapes needs to be tested to explore more extreme scenarios including agricultural abandonment and woody encroachment.

Acknowledgements

We thank Audrey Orcel, Stéphanie Périquet, Marion Salvi and Francesco de Bello for data collection, the Ecrins National Park for support, Villar d'Arène farmers for access to fields and interviews, the Joseph Fourier Alpine Station for access to facilities, and Philippe Choler, Fabien Quétier and Ulrike Tappeiner for insightful comments on the manuscript. This research was conducted on the long-term research site Zone Atelier Alpes, a member of the ILTER-Europe network. ZAA publication n° 002, with funding from projects ACI-ECCO ECOGER DIVHERBE and ANR BiodivERsA VITAL.

References

Albert, C.H., Thuiller, W., Yoccoz, N.G., Soudant, A., Boucher, F., Saccone, P. & Lavorel, S. (2010) Intraspecific functional variability: extent, structure and sources of variation within a French alpine catchment. *Journal of Ecology,* **98,** 604-613.

Andrieu, N., Josien, E. & Duru, M. (2007) Relationships between diversity of grassland vegetation, field characteristics and land use management practices assessed at the farm level. *Agriculture, Ecosystems and Environment,* **120,** 359-369.

Ansquer, P., Duru, M., Theau, J.P. & Cruz, P. (2009) Functional traits as indicators of fodder provision over a short time scale in species-rich grasslands. *Annals of Botany,* **103,** 117-126.

Bakker, M.M., Govers, G., Van Doorn, A., Quétier, F., Chouvardas, D. & Rounsevell, M.D.A. (2008) The response of soil erosion and sediment export to land-use change in four areas of Europe: The importance of landscape pattern. *Geomorphology,* **98,** 213-226.

Baraloto, C., Paine, C.E.T., Patiño, S., Bonal, D., Hérault, B. & Chave, J. (2009) Functional trait variation and sampling strategies in species-rich plant communities. *Functional Ecology,* **24,** 208-216.

Bennett, E.M., Peterson, G.D. & Gordon, L.J. (2009) Understanding relationships among multiple ecosystem services. *Ecology Letters,* **12,** 1394-1404.

Bruun, H.H., Fritzboger, B., Rindel, P.O. & Lund Hanssen, U. (2001) Species richness in grasslands: the relative importance of contemporary environment and land-use history since the Iron Age. *Ecography,* **24,** 569-578.

Bryan, B.A., Raymond, C.M., Crossman, N.D. & Macdonald, D.H. (2010) Targeting the management of ecosystem services based on social values: Where, what, and how? *Landscape and Urban Planning,* **97,** 111-122.

Carpenter, S.R., Mooney, H.A., Agard, J., Capistrano, D., De Fries, R.S., Diaz, S., Dietz, T., Duraiappah, A.K., Oteng-Yeboah, A., Pereira, H.M., Perrings, C., Reid, W.V., sarukhan, J., Scholes, R.J. & Whyte, A. (2009) Science for managing ecosystem services: Beyond the Millennium Ecosystem Assessment. *Proceedings of the National Academy of Sciences,* **106,** 1305-1312.

Chan, K.M.A., Shaw, M.R., Cameron, D.R., Underwood, E.C. & Daily, G.C. (2007) Conservation planning for ecosystem services. *PLoS Biology,* **4,** e379.

Chave, J., Coomes, D., Jansen, S., Lewis, S.L., Swenson, N.G. & Zanne, A.E. (2009) Towards a worldwide wood economics spectrum. *Ecology Letters,* **12,** 351-366.

Cornelissen, J.H.C., Lavorel, S., Garnier, E., Díaz, S., Buchmann, N., Gurvich, D.E., Reich, P.B., ter Steege, H., Morgan, H.D., van der Heijden, M.G.A., Pausas, J.G. & Poorter, H. (2003) Handbook of protocols for standardised and easy measurement of plant functional traits worldwide. *Australian Journal of Botany,* **51,** 335-380.

Cornelissen, J.H.C., Pérez-Harguindeguy, N., Díaz, S., Grime, J.P., Marzano, B., Cabido, M., Vendramini, F. & Cerabolini, B. (1999) Leaf structure and defence control litter decomposition rate across species and life forms in regional flora on two continents. *New Phytologist,* **143,** 191-200.

De Bello, F., Lavorel, S., Díaz, S., Harrington, R., Bardgett, R., Berg, M., Cipriotti, P., Cornelissen, H., Feld, C., Hering, D., Silva, P.M.d., Potts, S., Sandin, L., Sousa, J.P., Storkey, J. & Wardle, D. (2010) Functional traits underlie the delivery of ecosystem services across different trophic levels. *Biodiversity and Conservation,* **in press**.

De Chazal, J., Quétier, F., Lavorel, S., Van Doorn, A. & Castro, H. (2008) Including multiple differing stakeholder values into vulnerability assessments of socio-ecological systems. *Global Environmental Change,* **18,** 508-520.

de Groot, R.S., Alkemade, R., Braat, L., Hein, L. & Willemen, L. (2009) Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. *Ecological Complexity,* **in press**.

Di Rienzo, J.A., Casanoves, F. & Pla, L. (2008) f-Diversity. Cordoba, Argentina.

Diaz, S., Hodgson, J.G., Thompson, K., Cabido, M., Cornelissen, J.H.C., Jalili, A., Montserrat-Martí, G., Grime, J.P., Zarrinkamar, F., Astri, Y., Band, S.R., Basconcelo, S., Castro-Díez, P., Funes, G., Hamzehee, B., Koshnevi, M., Pérez-Harguindeguy, N., Pérez-Rontomé, M.C., Shirvany, F.A., Vendramini, F., Yazdani, S., Abbas-Azimi, R., Bogaard, A., Boustani, S., Charles, M., Dehghan, M., Torres-Espuny, d., Falczuk, V., Guerrero-Campo, J., Hynd, A., Jones, G., Kowsary, E., Kazemi-Saeed, F., Maestro-Martinez, M., Romo-Diez, A., Shaw, S., Siavash, B., Villar-Salvador, P. & Zak, M. (2004) The plant traits that drive ecosystems: Evidence from three continents. *Journal of Vegetation Science,* **15,** 295-304.

Diaz, S., Lavorel, S., De Bello, F., Quétier, F., Grigulis, K. & Robson, T.M. (2007) Incorporating plant functional diversity effects in ecosystem service assessments. *Proceedings of the National Academy of Sciences,* **104,** 20684-20689.

Egoh, B., Reyers, B., Rouget, M., Bode, M. & Richardson, D.M. (2009) Spatial congruence between biodiversity and ecosystem services in South Africa. *Biological Conservation,* **142,** 553-562.

Eigenbrod, F., Armsworth, P.R., Anderson, B.J., Heinemeyer, A., Gillings, S., Roy, D.B., Thomas, C.D. & Gaston, K.J. (2010) The impact of proxy-based methods on mapping the distribution of ecosystem services. *Journal of Applied Ecology,* **47,** 377-385.

Fortunel, C., Garnier, E., Joffre, R., Kazakou, E., Quested, H., Grigulis, K., Lavorel, S. & consortium, V. (2009) Plant functional traits capture the effects of land use change and climate on litter decomposability of herbaceous communities in Europe and Israel. *Ecology,* **90,** 598-611.

Fraterrigo, J.M., Turner, M.G. & Pearson, S.M. (2006) Interactions between past land use, life-history traits and understory spatial heterogeneity. *Landscape Ecology,* **21,** 777-790.

Garnier, E., Cortez, J., Billès, G., Navas, M.-L., Roumet, C., Debussche, M., Laurent, G., Blanchard, A., Aubry, D., Bellmann, A., Neill, C. & Toussaint, J.-P. (2004) Plant functional markers capture ecosystem properties during secondary succession. *Ecology,* **85,** 2630-2637.

Garnier, E., Lavorel, S., Ansquer, P., Castro, H., Cruz, P., Dolezal, J., Eriksson, O., Fortunel, C., Freitas, H., Golodets, C., Grigulis, K., Jouany, C., Kazakou, E., Kigel, J., Kleyer, M., Lehsten, V., Leps, J., Meier, T., Pakeman, R., Papadimitriou, M., Papanastasis, V., Quested, H., Quétier, F., Robson, T.M., Roumet, C., Rusch, G., Skarpe, C., Sternberg, M., Theau, J.P., Thébault, A., Vile, D. & Zarovali, M.P. (2007) A standardized methodology to assess the effects of land use change on plant traits, communities and ecosystem functioning in grasslands. *Annals of Botany,* **99,** 967-985.

Gartner, T.B. & Cardon, Z.G. (2004) Decomposition dynamics in mixed-species leaf litter. *Oikos,* **104,** 230-246.

Gaucherand, S. & Lavorel, S. (2007) A new protocol for a quick survey of functional traits values in a plant community. *Austral Ecology,* **32,** 927-936.

Gellrich, M. & Zimmermann, N.E. (2007) Investigating the regional-scale pattern of agricultural land abandonment in the Swiss mountains: A spatial statistical modelling approach. *Landscape and Urban Planninng,* **79,** 65-76.

Gimona, A. & van der Horst, D. (2007) Mapping hotspots of multiple landscape functions: a case study on farmland afforestation in Scotland. *Landscape Ecology,* **22,** 1255-1264.

Girel, J., Quetier, F., Bignon, A. & Aubert, S. (2010) *Histoire de l'Agriculture en Oisans.* Station Alpine Joseph Fourier, Grenoble, France.

Grace, J.B. (2006) *Structural Equation Modeling and Natural Systems.* Crambridge University Press.

Grêt-Regamey, A., Bebi, P., Bishop, I.D. & Schmid, W.A. (2008) Linking GIS-based models to value ecosystem services in an Alpine region. *Journal of Environmental Management,* **89,** 197-208.

Grime, J.P. (1977) Evidence for the existence of three primary strategies in plants and its relevance to ecological and evolutionary theory. *The American Naturalist,* **111,** 1169-1194.

Grime, J.P. (1998) Benefits of plant diversity to ecosystems: immediate, filter and founder effects. *Journal of Ecology,* **86,** 902-906.

Gross, N., Robson, T.M., Lavorel, S., Albert, C., Le Bagousse-Pinguet, Y. & Guillemin, R. (2008) Plant response traits mediate the effects of subalpine grasslands on soil moisture. *New Phytologist,* **180,** 652–662.

Gross, N., Suding, K.N. & Lavorel, S. (2007) Leaf dry matter content and lateral spread predict response to land-use change factors for six dominant species in subalpine grasslands. *Journal of Vegetation Science,* **18,** 289-300.

Gross, N., Suding, K.N., Roumet, C. & Lavorel, S. (2007) Complementarity as a mechanism of coexistence among co-dominant functional groups of graminoids. *Journal of Ecology,* **95,** 1296-1305.

Kattge, J., Ogle, K., Bönisch, G., Díaz, S., Lavorel, S., Madin, J., Nadrowski, K., Noellert, S., Sartor, K. & Wirth, C. (2010) A generic structure for plant trait databases. *Methods in Ecology and Evolution*, **in press**.

Kazakou, E., Vile, D., Shipley, B., Gallet, C. & Garnier, E. (2006) Co-variations in litter decomposition, leaf traits and plant growth in species from a Mediterranean old-field succession. *Functional Ecology,* **20,** 21-30.

Kienast, F., Bolliger, J., Potschin, M., Groot, R.S.d., Verburg, P.H., Heller, I., Wascher, D. & Haines-Young, R. (2009) Assessing Landscape Functions with Broad-Scale Environmental Data: Insights Gained from a Prototype Development for Europe. *Environmental Management,* **44,** 1099–1120.

Kleyer, M., Bekker, R.M., Knevel, I.C., Bakker, J.P., Thompson, K., Sonnenschein, M., Poschlod, P., van Groenendael, J.M., Klimeš, L., Klimešová, J., Klotz, S., Rusch, G.M., Hermy, M., Adriaens, D., Boedeltje, G., Bossuyt, B., Dannemann, A., Endels, P., Götzenberger, L., Hodgson, J.G., Jackel, A.K., Kühn, I., Kunzmann, D., Ozinga, W.A., Römermann, C., Stadler, M., Schlegelmilch, J., Steendam, H.J., Tackenberg, O., Wilmann, B., Cornelissen, J.H.C., Eriksson, O., Garnier, E. & Peco, B. (2008) The LEDA Traitbase: a database of life-history traits of the Northwest European flora. *Journal of Ecology,* **96,** 1266-1274.

Koch, E.W., Barbier, E.B., Silliman, B.R., Reed, D.J., Perillo, G.M., Hacker, S.D., Granek, E.F., Primavera, J.H., Muthiga, N., Polasky, S., Halpern, B.S., Kennedy, C.J., Kappel, C.V. & Wolanski, E. (2009) Non-linearity in ecosystem services: temporal and spatial variability in coastal protection. *Frontiers in Ecology and the Environment,* **7,** 29-37.

Kremen, C. (2005) Managing ecosystem services: what do we need to know about their ecology? *Ecology Letters,* **8,** 468-479.

Lavorel, S., Gachet, S., Sahl, A., Colace, M.-P., Gaucherand, S., Burylo, M. & Bonet, R. (2009) A plant functional traits data base for the Alps - Understanding functional effects of changed grassland management *Data mining for global trends in mountain biodiversity* (eds C. Körner & E. Spehn), pp. 106-123. CRC Press / /Taylor and Francis, Boca Raton.

Lavorel, S. & Garnier, E. (2002) Predicting the effects of environmental changes on plant community composition and ecosystem functioning: revisiting the Holy Grail. *Functional Ecology,* **16,** 545-556.

Lavorel, S., Grigulis, K., McIntyre, S., Garden, D., Williams, N., Dorrough, J., Berman, S., Quétier, F., Thébault, A. & Bonis, A. (2008) Assessing functional diversity in the field – methodology matters! *Functional Ecology,* **22,** 134-147.

Leps, J., De Bello, F., Lavorel, S. & Berman, S. (2006) Quantifying and interpreting functional diversity of natural communities: practical considerations matter. *Preslia,* **78,** 481-501.

Martin, G., Hossard, L., Theau, J.P., Therond, O., Josien, E., Cruz, P., Rellier, J.P., Martin-Clouaire, R. & Duru, M. (2009) Characterizing potential flexibility in grassland use - An application to the French Aubrac region. *Agronomy for Sustainable Development,* **29,** 381–389.

Mason, N.W.H., MacGillivray, K., Steel, J.B. & Wilson, J.B. (2003) An index of functional diversity. *Journal of Vegetation Science,* **14,** 571-578.

Maurer, K., Weyand, A., Fischer, M. & Stöcklin, J. (2006) Old cultural traditions, in addition to land use and topography, are shaping plant diversity of grasslands in the Alps. *Biological Conservation,* **130,** 438-446.

Millennium Ecosystem Assessment (2005) *Ecosystems and human well-being. Synthesis. A report of the Millennium Ecosystem Assessment.* Island Press, Washington.

Metzger, M.J., Rounsevell, M.D.A., Acosta-Michlik, L., Leemans, R. & Schröter, D. (2006) The vulnerability of ecosystem services to land use change. *Agriculture, Ecosystems & Environment,* **114,** 69-85.

Mokany, K., Ash, J. & Roxburgh, S. (2008) Functional identity is more important than diversity in influencing ecosystem processes in a temperate native grassland. *Journal of Ecology,* **96,** 884-893.

Mottet, A., Ladet, S., Coqué, N. & Gibon, A. (2006) Agricultural land-use change and its drivers in mountain landscapes: A case study in the Pyrenees. *Agriculture, Ecosystems & Environment,* **114,** 296-310.

Naidoo, R., Balmford, A., Costanza, R., Fisher, B., Green, R.E., Lehner, B., Malcolm, T.R. & Ricketts, T.H. (2008) Global mapping of ecosystem services and conservation priorities. *Proceedings of the National Academy of Sciences,* **105,** 9495-9500.

Naidoo, R. & Ricketts, T.H. (2006) Mapping the Economic Costs and Benefits of Conservation. *PLoS Biology,* **4,** e360

Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, D., Chan, K.M., Daily, G.C., Goldstein, J., Kareiva, P.M., Lonsdorf, E., Naidoo, R., Ricketts, T.H. & Shaw, M. (2009) Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. *Frontiers in Ecology and the Environment,* **7,** 4-11.

Osty, P. (1971) Influence of soil conditions on its moisture at pF 3. *Annales Agronomiques,* **2,** 451-476.

Petchey, O.L. & Gaston, K.J. (2006) Functional diversity: back to basics and looking forward. *Ecology Letters,* **9,** 741-748.

Pontes Da Silva, L., Soussana, J.F., Louault, F., Andueza, D. & Carrère, P. (2007) Leaf traits affect the above-ground productivity and quality of grasses. *Functional Ecology,* **21,** 844-853.

Quétier, F., Lavorel, S., Daigney, S. & De Chazal, J. (2009) Assessing ecological and social uncertainty in the evaluation of land-use impacts on ecosystem services. *Journal of Land Use Science,* **4,** 173-199.

Quétier, F., Lavorel, S., Thuiller, W. & Davies, I.D. (2007) Plant trait-based assessment of ecosystem service sensitivity to land-use change in mountain grasslands. *Ecological Applications,* **17,** 2377–2386.

Quétier, F., Rivoal, F., Marty, P., De Chazal, J. & Lavorel, S. (2010) Social representations of an alpine grassland landscape and socio-political discourses on rural development. *Regional Environmental Change,* **10**, 119-130.

Quétier, F., Thébault, A. & Lavorel, S. (2007) Linking vegetation and ecosystem response to complex past and present land use changes using plant traits and a multiple stable state framework. *Ecological Monographs,* **77,** 33-52.

Reyers, B., O'Farrell, P.J., Cowling, R.M., Egoh, B.N., Maitre, D.C.L. & Vlok, J.H.J. (2009) Ecosystem Services, Land-Cover Change, and Stakeholders: Finding a Sustainable Foothold for a Semiarid Biodiversity Hotspot. *Ecology and Society,* **14,** 38. [online].

Robson, T.M., Lavorel, S., Clément, J.C. & Le Roux, X. (2007) Neglect of mowing and manuring leads to slower nitrogen cycling in subalpine grasslands. *Soil Biology and Biogeochemistry,* **39,** 930-941.

Scherer-Lorenzen, M. (2008) Functional diversity affects decomposition processes in experimental grasslands. *Functional Ecology,* **22,** 547-555.

Schumacher, J. & Roscher, C. (2009) Differential effects of functional traits on aboveground biomass in semi-natural grasslands. *Oikos,* **118,** 1659-1668.

Stewart, K.E.J., Bourn, N.A.D. & Thomas, J.A. (2001) An evaluation of three quick methods commonly used to assess sward height in ecology. *Journal of Applied Ecology,* **38,** 1148-1154.

Suding, K.N. & Goldstein, L.J. (2008) Testing the Holy Grail framework: using functional traits to predict ecosystem change. *New Phytologist,* **180,** 559-562.

Turner, B.L. II. (2010) Sustainability and forest transitions in the southern Yucatán: The land architecture approach. *Land Use Policy,* **27,** 170-179.

Ustin, S.L. & Gamon, J.A. (2010) Remote sensing of plant functional types. *New Phytologist,* **186,** 795-816.

Venables, W.N. & Ripley, B.D. (2002) *Modern Applied Statistics with S.* Springer-Verlag.

Verburg, P.H., van der Steeg, J., Veldkamp, A. & Willemen, L. (2009) From land cover change to land function dynamics: A major challenge to improve land characterization. *Journal of Environmental Management,* **90,** 1326-1335.

Vile, D., Shipley, B. & Garnier, E. (2006) Ecosystem productivity relates to species' potential relative growth rate: a field test and a conceptual framework. *Ecology Letters,* **9,** 1061-1067.

Violle, C., Navas, M.L., Vile, D., Kazakou, E., Fortunel, C., Hummel, I. & Garnier, E. (2007) Let the concept of trait be functional! *Oikos,* **116,** 882-892.

Vojtech, E., Loreau, M., Yachi, S., Spehn, E.M. & Hector, A. (2008) Light partitioning in experimental grass communities. *Oikos,* **117,** 1351-1361.

Westoby, M. (1998) A leaf-height-seed (LHS) plant ecology strategy scheme. *Plant and Soil,* **199,** 213-227.

Willemen, L., Hein, L., van Mensvoort, M.E.F. & Verburg, P.H. (2010) Space for people, plants, and livestock? Quantifying interactions among multiple landscape functions in a Dutch rural region. *Ecological Indicators,* **10,** 62-73.

Synthèse partie I

Cette première partie a exploré les relations entre les écosystèmes et les services écosystémiques et avait pour but de répondre à deux questions : (1) Quels services écosystémiques sont perçus, utilisés et/ou appréciés par les *stakeholders* ? ; (2) Quels services écosystémiques sont potentiellement fournis par les prairies étant donné les dynamiques écologiques ?

Le concept de service des écosystèmes est utilisé depuis 2005 dans de nombreuses disciplines scientifiques et commence à être largement adopté dans les domaines politique et entrepreneurial. Il traduit l'idée que les sociétés humaines tirent divers bénéfices, directs et indirects, des écosystèmes (par exemple, la production agricole, l'esthétique ou la régulation du climat). Malgré de nombreux points communs, plusieurs définitions et usages du concept de service des écosystèmes coexistent aujourd'hui, en parallèle d'autres termes proches tels que les services écologiques, environnementaux ou du paysage. Cette variété terminologique traduit des différences de compréhension du concept mais aussi des différences d'évaluation de la qualité, quantité ou localisation de la fourniture de services. Cette variété est susceptible de compliquer la communication et l'utilisation du concept pour la conservation de la nature et la gestion des ressources naturelles.

Afin d'illustrer cette variété et son implication sur les études des services des écosystèmes, j'ai passé en revue la manière dont ces derniers sont abordés dans les principales publications scientifiques sur ce sujet et, ensuite, replacé ces définitions et usages dans la structure conceptuelle publiée par Haines-Young et Potschin (2010) (Chapitre 2). Celle-ci se présente sous la forme d'une cascade (Figure 1) partant des composantes biophysiques de l'écosystème (par ex. les espèces et leurs abondances) pour aboutir aux bénéfices que l'homme retire des écosystèmes, moyennant dans certains cas l'apport de capital humain (machines, travail, infrastructures, …). Les services des écosystèmes se positionnent entre ces deux composantes de la cascade distinguant, d'une part, la fourniture potentielle de services qui renvoie à la notion de fonction des écosystèmes *(ecosystem function* en anglais) et, d'autre part, les services des écosystèmes *sensu stricto* (*ecosystem services*) qui sont utilisés, consommés ou appréciés par l'homme. La confusion la plus fréquente dans l'utilisation du concept concerne la distinction entre ces deux aspects d'offre et de demande des services écosystémiques. La plupart des études se focalisent sur la fourniture potentielle de services (*ecosystem functions*) en prenant en compte des paramètres écologiques ou des variables de substitution (*proxys*), mais rarement la perception de ces services par les bénéficiaires (*ecosystem services*) et l'importance qu'ils leur accordent.

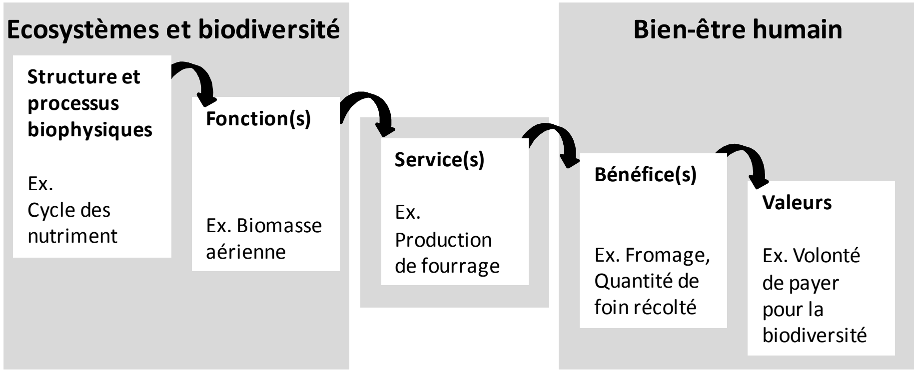


Figure 1 : Cascade conceptuelle illustrée avec l'exemple des prairies subalpines (chapitre 2)

Afin de combler ce manque de connaissances sur la demande en terme de services et d'identifier les principaux services liés aux zones agricoles de montagne, j'ai réalisé des entretiens semi-dirigés auprès de 13 experts régionaux issus de différents secteurs d'activité (agriculture, conservation, tourisme, développement rural) mais aussi de 6 représentants d'agriculteurs et d'habitants de Villar d'Arène (Chapitre 3). Cette étude a révélé que le concept de services des écosystèmes n'était généralement pas connu des interviewés. Seulement la moitié des professionnels avaient déjà entendu parler du concept et seulement la moitié d'entre eux ont proposé une définition correcte. Ceci confirme d'un côté le besoin, identifié dans la bibliographie, de clarification des définitions autour de la notion de SE, mais aussi les risques associés à l'application d'un concept flou. Néanmoins, après explications et définitions de 21 services pré-listés d'après la bibliographie, ce concept a été rapidement assimilé par les acteurs locaux (*stakeholders)* puisque ceux-ci ont pu dégager, un jeu de services qu'ils considèrent comme importants. Dans ce jeu figurent par exemple la stabilité du sol, la quantité d'eau, la qualité et la quantité du fourrage, la conservation de la diversité botanique, l'esthétique et les activités récréatives. Des différences de perceptions des services des écosystèmes au sein du site d'étude (entre les *stakeholders*) ont cependant été mises en évidence. Les agriculteurs et les experts de Villar d'Arène ne considèrent simultanément comme important que trois services : qualité et quantité du fourrage et conservation de la diversité floristique. De leur côté, les agriculteurs ont aussi considéré comme importants la fertilité du sol, l'humidité du sol, le contrôle des ravageurs et l'esthétique, alors que les experts régionaux ont pointé la pollinisation, la stabilité du sol, la présence d'habitat pour la faune, la régulation des risques naturels, la qualité de l'eau, la séquestration du carbone, les activités récréatives et le sentiment d'appartenance (*sense of place*). Ces différences reflètent les connaissances techniques et locales et les objectifs propres à chacun des deux groupes d'acteurs.

Un stage que j'ai encadré dans le cadre de cette thèse (Grard, 2010) a montré que les services écosystémiques les plus fréquents dans les politiques agricoles, environnementales et territoriales françaises sont la qualité de l'eau, l'esthétique, les activités récréatives, la conservation de la biodiversité, la quantité d'eau et la réduction des risques naturels (résultats non présentés dans les chapitres de cette thèse). Sur cette base, j'ai réalisé une enquête auprès de 90 touristes durant l'été 2011 sur la commune de Villar d'Arène afin d'identifier les éléments paysagers et floristiques contribuant à l'esthétique des prairies. Les résultats (non présentés dans les chapitre de cette thèse) montrent que 54% des enquêtés considèrent que les prairies de fauche hors terrasses sont les plus esthétiques (Figure 2) considérant les couleurs des fleurs et la diversité comme les principaux critères contribuant à leur beauté.

Figure 2 : Photos présentées aux touristes et résultats de leur classification par ordre de préférence selon l'esthétisme attribué aux prairies. Classement de la prairie considérée comme la plus esthétique (1) à celle considérée comme la moins esthétique (6), selon la moyenne des scores attribués à chacune des photos par les 90 enquêtes réalisées.

A ce stade, mes résultats suggèrent donc qu'une bonne compréhension du concept de services ainsi que des définitions claires sur son utilisation doivent être établies préalablement à toute étude sur ce sujet. De plus, pour veiller à la pertinence des études par rapport aux problématiques locales, il semble important d'associer les *stakeholders* à l'identification des principaux services des écosystèmes. Toujours dans un souci de pertinence par rapport aux attentes et aux besoins des *stakeholders*, les études doivent se concentrer sur les services jugés importants d'après leurs indications.

Conformément à l'approche conceptuelle en cascade (chapitre 2), il s'agit ensuite d'identifier à quelle(s) fonction(s) des écosystèmes est associé chacun de ces services jugés importants. Ces fonctions des écosystèmes ont été quantifiées et spatialisées à l'aide des modèles statistiques basés sur des données de traits fonctionnels végétaux et microbiens[10], de composantes abiotiques et d'utilisation du sol (Figure 3).

[10] Les services liées aux traits microbiens (fertilité du sol et qualité de l'eau) ont été modélisés postérieurement à la publication du chapitre 4 et sont présentés uniquement dans le chapitre 6.

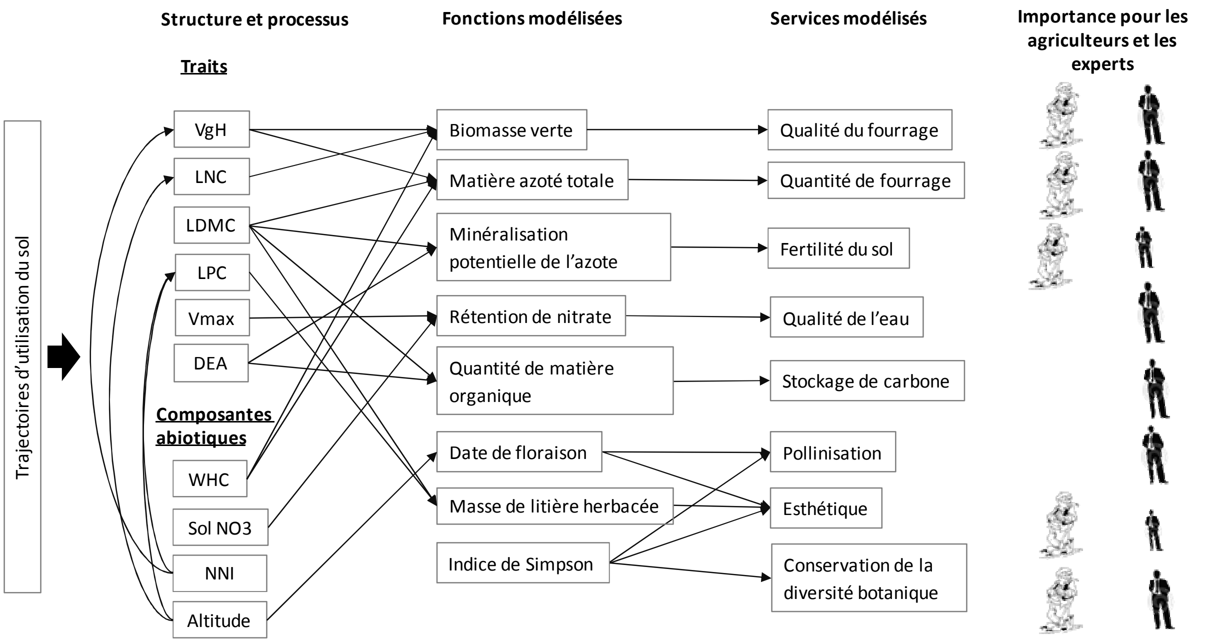


Figure 3 : Positionnement des données collectées et modélisées selon la cascade de Haines-Young et Potschin (2010) afin d'étudier les services des écosystèmes sur la commune de Villar d'Arène. VgH, Hateur de végétation ; LNC, contenu en azote des feuilles ; LDMC, contenu en matière sèche des feuilles ; LPC, contenu en phosphore des feuilles; Vmax, constante de l'activité de l'enzyme nitrifiante; DEA, Activité de l'enzyme denitrifiante ; WHC, Capacité de rétention en eau ; Sol NO3, teneur en azote du sol ; NNI, indice de nutrification azoté.

Cette nouvelle approche de quantification, de spatialisation et de compréhension de la fourniture de multiples services dans le paysage basée sur les relations entre les traits fonctionnels, les composantes abiotiques et la gestion des prairies apporte plusieurs intérêts par rapport aux méthodes classiques prenant en compte uniquement l'utilisation du sol (Chapitre 4). Son intérêt principal est qu'elle permet l'étude des services et de leurs variations en réponse à des facteurs environnementaux à une échelle géographique fine, sur un territoire d'étude marqué par une seule catégorie d'utilisation du sol (prairies permanentes). Dans ce cas particulier, il est important de prendre en considération les variations des facteurs environnementaux (modalités de gestion, altitude, climat, propriétés du sol, …) pour expliquer les variations de services des écosystèmes à l'échelle du territoire étudié. Dans cette nouvelle approche, les traits fonctionnels constituent la charnière entre les facteurs environnementaux et les propriétés des écosystèmes.

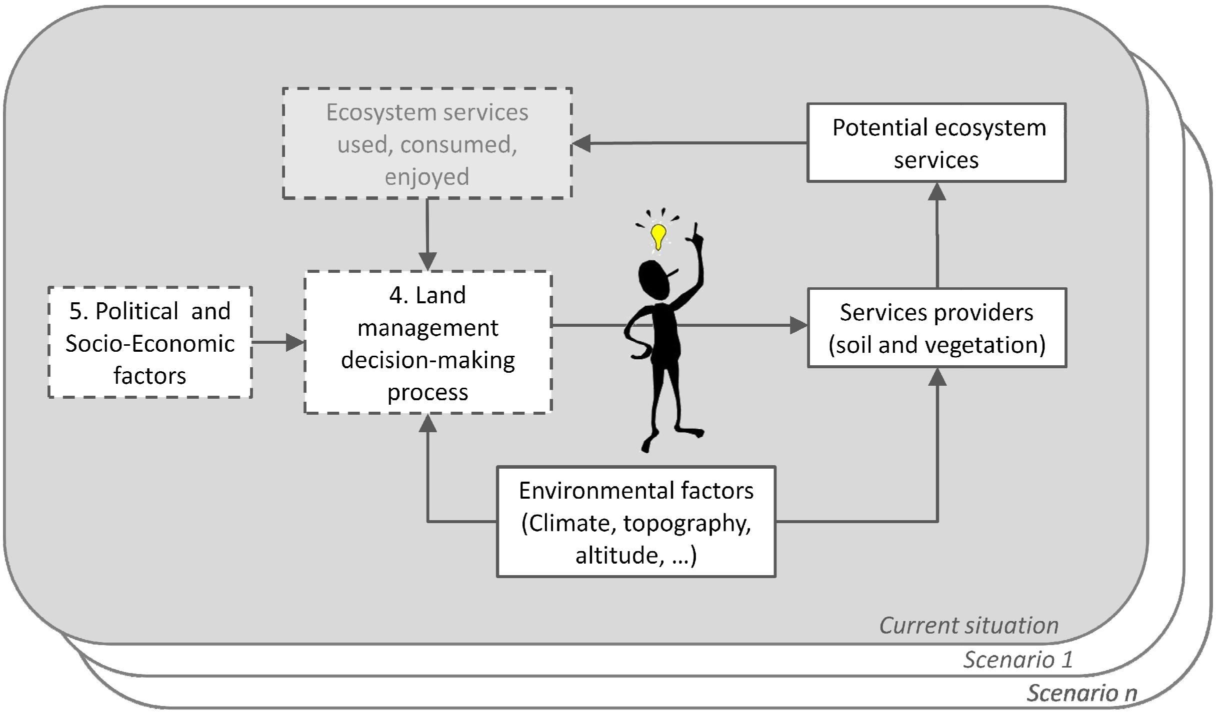
En plus d'expliquer les variations individuelles de services, cette nouvelle approche permet aussi d'améliorer la compréhension des mécanismes écologiques qui sous-tendent la fourniture de « bouquets » de services par les écosystèmes. C'est ainsi que la cartographie des services des écosystèmes montre par exemple que : (1) les prés de fauche fertilisés et les alpages fournissent le plus de services alors que les prairies non fauchées sont particulièrement peu favorables, (2) il est possible de concilier conservation de la biodiversité et les valeurs culturelles associées avec la valeur agronomique car les traits fonctionnels qui y contribuent sont orthogonaux (traits foliaires et hauteur de végétation).

L'approche par les traits fonctionnels trouve par ailleurs un écho particulier dans les descriptions fournies par les interviewés, notamment les agriculteurs, qui décrivent souvent les relations entre services des écosystèmes et biodiversité plus en terme de traits des plantes (ex. feuilles larges, herbe vert foncé) qu'en terme d'espèces (Chapitre 3). Les interviewés ont généralement perçu des liens de causalité entre les pratiques agricoles et les services, par contre ils ont éprouvé plus de difficulté à percevoir des relations entre services. Pourtant, notre modélisation a mis en avant certains compromis important à prendre en compte pour une gestion durable des services des écosystèmes (ex. quantité/qualité du fourrage, quantité/stockage carbone dans les prairies à *Festuca paniculata*) (Chapitre 4). Ce constat m' a amenée à approfondir, dans la Partie II de cette thèse, les relations entre services notamment en cas de changement de gestion des terres et de poursuivre notre démarche participative afin de discuter avec les éleveurs de l'effet de changement de pratiques sur la fourniture de services des écosystèmes.

Partie II

De l'homme à l'écosystème

Evolutions futures



Chapitre 5

Une approche participative pour comprendre l'adaptation des exploitations aux sécheresses et changements socio-politiques et construire de scénarios de gestion des prairies[11]

Abstract

Mountain grassland ecosystems are particularly vulnerable to direct changes in climate and indirect changes through farmers adaptive management in the face of changes in climatic climate and socio-economic context and policies. We modeled expected spatio-temporal trajectories of land management of a mountain grassland landscape in the French Alps under a range of climatic and socio-economic scenarios for 2030 which were constructed using an advanced participatory approach with a variety of stakeholders. First, regional experts from nature conservation and agricultural extension were involved in the co-development of detailed qualitative climatic and socio-economic scenarios, expressed as coherent storylines. Second, to translate these storylines into quantitative land management scenarios, we used a role playing game whereby local stakeholders were put in an imaginary future situation and asked to make decisions under scenarios constraints. For each scenario, game outcomes were integrated in the mapping of future land management at parcels to landscape scales. Main adaptations were conversion from mowing to grazing and increasing manured area, with varying proportions and locations for these two types of changes differing in across scenarios.

1 Introduction

Mountain ecosystems are highly vulnerable to climate change and extreme events (Engler et al., 2011) such as the increased occurrence of drought observed during the last decades (Bigot et al., 2010; Lemaire and Pflimlin, 2007)) and predicted in the future (IPCC, 2007). However, in subalpine grasslands climate effects on vegetation tend to be overridden by management effects

[11] Ce chapitre est l'objet d'un article soumis à Landscape and Urban planning : Pénélope Lamarque, Baptiste Nettier, Cécile Barnaud, Aloïs Artaux, Claire Eveilleau, Laurent Dobremez, Sandra Lavorel, A participatory approach to map land management change based on the adaptive management of mountain livestock systems to drought and socio-economic scenarios.

(e.g. grazing or mowing) (Benot et al., in revision; Lavorel, 2011; Vittoz et al., 2009). Mountain livestock farming is inherently sensitive to climate due to its reliance on grass production, requiring breeders and shepherds to adapt their systems to reduce vulnerability (Lemaire and Pflimlin, 2007). As land use and management adaptation is such an important component of climate change impacts, these indirect effects need to be considered in combination with direct climate effects when assessing climate change scenarios for mountain grassland ecosystems. Nevertheless, adaptations in agriculture are not influenced only by climatic stimuli, despite their importance. Non-climatic forces such as markets and subsidies, policies and societal values also play a significant role in agricultural decision-making (Bryant et al., 2000; Nettier et al., 2010; von Glasenapp and Thornton, 2011). Environmental and land-use scenarios have been developed and used to enhance understanding of the complex interactions and the dynamics of change of all these parameters (Moss et al., 2010).

Scenarios are « plausible and often simplified descriptions of how the future may develop based on a coherent and internally consistent set of assumptions about key driving forces and relationships » (Millennium Ecosystem Assessment, 2005). One can emphasize that «scenarios explore the possible, not just the probable, and challenge their users to think beyond what it sought as conventional wisdom » (Jäger et al., 2008). “Predictive” scenarios of the type “What-if” address the specific question: “What will happen, on the conditions of specific events?” (Borjeson *et al.* (2006), as we did in this study. Answers to this question can be attained by quantitative (e.g. simulation modelling) (e.g. Schroter et al., 2005) or qualitative (e.g. storylines) scenarios (e.g. SRES storylines) or both (e.g. Millennium Ecosystem Assessment, 2005; Walz et al., 2007). Each method has advantages and limits but qualitative scenarios allow to take into account more variability and uncertainty than quantitative model do (Coreau, 2009). Usually, scenario building follows a framework composed of five main steps: (1) defining the focal question, (2) identifying the key drivers, (3) determining the scenario logic, (4) describing scenario assumptions using qualitative scenario storylines, and (5) assessing scenario outcomes or developing quantitative scenarios based on numerical model (Metzger et al., 2010). It is now common to use a participatory approach in the development of qualitative scenarios, “meaning a set of procedures through which experts and stakeholders work together to develop the scenarios” (Alcamo et al., 2006). Stakeholders are usually involved in storylines development (step four of scenario building) or in some specific step of qualitative scenarios development (Bohunovsky et al., 2010) because quantitative knowledge is not needed (van Vliet et al., 2010) and storylines use a more understandable language to communicate with stakeholders about the future (Alcamo et al., 2006; van Vliet et al., 2010). Participatory approaches are particularly valuable at regional or local levels (Alcamo et al., 2006) because stakeholders are very likely to be actors themselves (Bohunovsky et al., 2010) with extensive local knowledge (Swetnam et al., 2011; Walz et al., 2007). Several methods are used in participatory scenario development such as interviews or focus groups, stakeholders panel workshops, gaming workshops, policy exercises, or story and simulation approaches (Alcamo et al., 2006). Nevertheless, it is essential to select appropriate tools to avoid some drawbacks such as the influence of personal judgement of storylines narratives (Metzger et al., 2010), limited credibility of scenarios by a potential lack of diversity among participants (Rounsevell and Metzger, 2010), or the compromise between creativity and methodological structure (van Vliet et al., 2010). A challenge for participatory development of future scenarios is the integration of qualitative and quantitative analysis, the latter being frequently impeded by the difficulties met to translate qualitative assumptions of driving forces into quantitative model input (Volkery et al., 2008; Walz et al., 2007). However, Swetnam et al. (2011) suggested that these difficulties can be overcome by defining quantitative rules during stakeholder workshops for translating qualitative scenarios into quantitative maps.

This study aimed at testing a methodology for downscaling climate and socio-economic scenarios into land management maps. For that purpose, we used a participatory scenario approach combining qualitative storylines and quantitative analysis. The framework, described step by step hereafter, is composed of two main phases. The first phase is the co-development of highly detailed qualitative storylines with regional experts who share local knowledge. The second phase explores with farmers how these scenarios could affect land management. In this phase, a role playing game was used to involve farmers in the downscaling of scenarios to land management maps, and to capture their decisions (Martin et al., 2011; Pak and Brieva, 2010; Washington-Ottombre et al., 2010). The separation in two phases allows to integrate driving forces and the land-use decisions underpinning land-use change in our analyses.

To make sure that drivers of change are relevant at regional level, in the first step storylines were developed simultaneously for two mountain sites, the Vercors Regional Park and the Villar d'Arène municipality in Ecrins National Park. The methodology for the second step was tested only on Villar d'Arène where the entire farmed area is composed of grasslands managed by only eight farmers, allowing us to downscale the scenarios based on highly detailed data and local knowledge. Site description and results concern the Lautaret only because presentation of both sites would have yielded a substantial amount of data and also because the aim of this paper is methodological.

2 Study area

The study site (45°03' N, 6°24' E) is located in the Central French Alps Villar d'Arène, a municipality of 77,5 km² and 271 inhabitants . Villar d'Arène is a member of the Ecrins National Park and hostes a Long Term Socio-Ecological Research platform (LTSER) where detailed data on vegetation, climate and farming systems have been collected for a decade. The total area of the south-facing slopes used for livestock production is 13 km² and the elevation ranges from 1552 to 2500 m a.s.l. Climate is subalpine with a strong continental influence due to a rain shadow with respect to dominant westerly winds. Mean annual rainfall is 956 mm and the mean monthly temperatures of -4.6°C in January to 11°C in July (at 2050 m above sea level). Rainfall occurs mainly during the cooler months, with 40% of annual rainfall during the growing season (April-September). The alpine meadows above 2200 m have been grazed extensively in summer for centuries, but the lower slopes have undergone land use change over the last century. Following rural exodus at the beginning of the 20th century, former arable land on terraced slopes (1650-2000 m) was converted into grasslands that are now grazed or mown where they are accessible to machinery (Girel et al., 2010). These recent grasslands and those used for hay production since the 1700s (1800-2200 m) (Table 1), have been managed at low intensity, with low stocking rates, very low manure inputs (every two or three years) and a single annual hay cut. Farmers try to be fodder self-sufficient because winter season (6 to 7 months) and also because of the cost of fodder. Eight farmers remain today, of which five farm full-time, one farms part-time and two are retired but continue to farm. Usually one member of the household works outside of the farm (Deboeuf, 2009). The eight farms can be classified into three categories according to their production systems: (1) lamb production (3 farms, mean = 21 livestock units (LU), 19 ha); (2) production of calves and heifers (3 farms, mean = 67 LU, 55 ha), (3) mixed sheep and cattle production (2 farms, mean = 54 LU, 48 ha). During summer, the alpine meadows are managed by a shepherd keeping track of the local sheep herds along with his own sheep (around 1500 sheep). Only two farmers sell a part of their production by direct sales. The region falls within European Less Favoured Areas, where European subsidies constitute a significant share of farm income as a means to make up for low productivity. In this municipality as in some others in mountain areas (Bossy, 1985) agricultural parcels of the multiple landowners are pool into a communal organization (Association Foncière Pastorale (AFP)) which allocates parcels among farmers in order to distribute land more equitably according to technical constraints associated with the mountain terrain. In addition to agriculture, tourism is a dominant economic activity in the region, which is recognized for its aesthetic, cultural and conservation value (part of the Ecrins National Park) and offers opportunities for recreation (Quétier et al., 2010a). We will refer hereafter to land management instead of the commonly used but less detailed term of land-use, because the area is mainly composed of grassland used for livestock production.

Land management type	Altitude range (m)	Slope (°)	Fodder production (T/ha)
Terraces mown and manured	1584 - 1944	0,16 - 37	306,67 – 463
Terraces mown	1554 - 1938	1,12 -61,3	300 - 487
Terraces grazed	1539 - 1794	0,39 - 59	298, 3 – 485,8
Unterraced grasslands mown	1854 - 2013	1,65 - 34,2	357, 2 – 464, 1
Unterraced grasslands grazed	1702 - 2024	1,28 - 43,7	312,3 – 489, 8
Alpine meadows	2228 - 2710	0,33 - 63,5	316,5 – 418, 3

Table 1: Current characteristics of grasslands types of Villar d'Arène (minimum-maximum).

The main dataset used in this study is a land management map derived from different surveys (2003-5, 2009) with farmers and analysis of aerial photographs and cadastral maps since 1810 (Girel et al., 2010; Quétier et al., 2007b)). This map contains six land management classes (Table 1) at a resolution of 20 m and has been given a nominal date of 2009. We used additional spatial information: elevation and slope obtain from a 10 m Digital Elevation Model under ArcGiS 9.2, ESRI; settlements, farms and roads digitized from the 1:25000 topographic map (IGN); fodder production modelled according to altitude, soil parameters and plant functional traits (Lavorel et al., 2011). In addition to this spatial information we used results from a survey of farmers' responses to recent droughts, which showed, that farmers responded mainly by purchasing fodder to compensate for yield loss (Nettier et al., 2010). Finally we used on-site experimental data to quantify expected drought effects on vegetation composition and fodder production (Benot et al., in revision).

3 Research framework

3.1 Framework

The step-wise methodology presented hereafter aims at developing local climatic and socio-economic scenarios to understand adaptive management of farmers to changing contexts and to map land management changes at parcels, farms and landscape scales (Figure 1). First, regional experts from nature conservation and agricultural extension were involved in the co-development of detailed qualitative climatic and socio-economic scenarios, expressed as coherent storylines. Secondly, the storyline impacts were downscaled to land management using a role playing game whereby local farmers were put in an imaginary future situation and asked to make decisions under scenarios constraints. This participative approach involved a variety of stakeholders all along the framework (Figure 1).

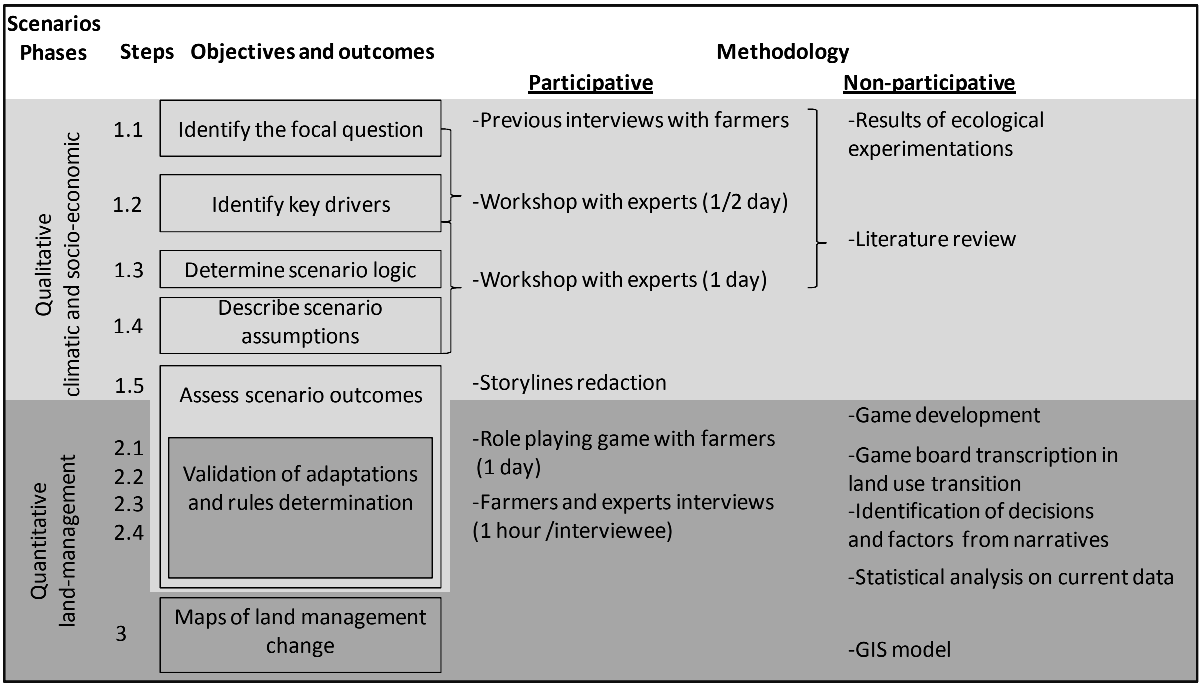


Figure 1: Framework of climatic and socio-economic scenarios (light grey) and land management scenarios (dark grey) building according to usual phases of scenario development (Metzger et al., 2010)

3.1.1 Co-construction with experts and storylines

The first four steps of the scenario development (Figure 1) aimed at building detailed and relevant storylines at regional scale. Previous work on the study area on farmers' adaptation to climate change showed that drought is only one factor among others and that socio-economic context also influences their decisions (Nettier et al., 2010). Therefore, participants and scientists agreed to construct four scenarios following a prospective approach (de Jouvenel, 2002) coupling two climate alternatives with two socio-economic alternatives. Socio-economic alternatives were downscaled from national and international scenarios (Agrimonde®, 2009; Millenium Ecosystem Assessment (MEA) Carpenter et al., 2005; Les Nouvelles Ruralités (NR) Mora, 2008): one reflected continued globalisation (see 'Global orchestration' in MEA, G0 in Agrimonde® and Scenario 1 in NR) and the other one explored an increase in regionalisation of policy and economics with flexible governance (see 'Adapting mosaïc' in MEA and scenario 4 in NR). Climate alternatives were formulated on the basis of downscaled projections of climate change models for 2050 (Boe et al., 2006; Pagé et al., 2008) and represented as a five year window.

Regional experts were involved in scenario development in order to draw contrasted and plausible alternatives, reflecting local features and being relevant to local stakeholders. Regional experts were chosen according to their expertise relevant to the scenario exercise (Alcamo et al., 2006) and were identified thanks to a previous work on actors network in the study area (Grard, 2010). Selection also ensured a diversity of knowledge by involving 10 stakeholders from land management and nature conservation bodies and 9 researchers in agronomy, forest science and ecology. Such a diversity of experts is important for sharing knowledge, but also to avoid a single vision of the future (Volkery et al., 2008) and to avoid the imprint of social contexts of origin in the cognitive style of scenarios (Garb et al., 2008). Co-construction with experts was conducted during two workshops (1/2 day + 1day) structured according the usual steps of scenario building (step 1.1 to 1.5 in figure 1) with additional individual interviews to refine storylines after the second workshop. In a concern to enhance scenario developer-user relations (Garb et al., 2008), particular attention was devoted to giving, along with general trends (e.g. % precipitation decrease) as in usual storylines, concrete scenario elements for presentation to farmers such as expected effects on vegetation composition, fodder production and water availability across the five scenario years (step 2.1 in Figure 1).

Discussions of the second workshop also anticipated possible adaptations based on recent responses to climate or economic events (Nettier et al., 2010) so as to ensure acceptability of scenarios to farmers and to avoid in so far as possible either setback or win-win scenarios.

3.1.2 Developing the role playing game

For allowing farmers to propose coherent adaptations, scenarios need to be presented in a manner that allows them to understand and to assimilate quickly their multiple parameters. Role-playing games can address this challenge by converting scientific concepts into usable forms for farmers (Barreteau et al., 2001; Martin et al., 2011). A role-playing game (RPG) is a model allowing to put players in a given situation with the help of different supports (Barnaud et al, 2008 , Economie Rurale). Our choice to use a RPG rather than more traditional methods such as semi-directive interviews or focus group was motivated by six objectives : primarily (1) to make scenario content easily understandable by farmers (Martin et al., 2011), (2) to develop a spatial support as a means to downscale land management change (Etienne, 2003; Rouan et al., 2010; Washington-Ottombre et al., 2010) according to the scenarios, (3) to identify decisions and factors that drive land management change (Pak and Brieva, 2010); and secondarily (4) to stimulate collective strategies and understand interactions among farmers (Barnaud et al., 2010) which are likely to be important in this municipality given the presence of the AFP collective mechanism (see section 2.1). In addition to these expected outcomes, RPG also allow (5) to elicit stakeholders' representation of the system (Castella et al., 2005) as players bring along their own habits and strategies to make choices within contexts given by the game (Naivinit, 2009), and (6) to share knowledge between scientists (drought effects on vegetation) and farmers (effects on farm and land management) (Voinov and Bousquet, 2010).

The game was designed by the scientists for only one type of actors: the eight farmers of the municipality, playing their own roles and starting from current conditions each with the land and herd of their managing system. The game was organized in two sessions corresponding to the two climate alternatives. The game in itself was a translation of the storylines of climate alternatives which described the impacts of drought events on the productivity of grasslands, while the socio-economic context remains as in current conditions.

Both sessions were divided into five rounds representing a year each. A major stake for livestock farmers is to manage to feed the herd during the vegetation season (May to November) and to produce enough stock of forage for the entire winter. Similar to other games such as the "Forage Rummy" (Martin et al., 2011), the aim for players was thus to successfully meet this challenge given grassland production under climate constraints presented for each year. In contrast with such other games, we incorporated the spatial characteristics of adaptations to obtain spatially explicit scenarios. This was done by using a two-dimensional board game composed of cells representing land-management types (e.g. mown and fertilized terraces or grazed unterraced grasslandss; see Table 1 for a full description) in the same proportions as in the actual landscape (Figure 2). This feature was considered essential to put farmers in a close to real situation and to make sure that game results could be translated into landscape maps for the site. The time step was a year (one round). On the board game, each player followed the rules defined at the beginning of each round to manage pieces representing cows, sheep, fodder and/or manure. Given the annual time step, and that most fields can only be used once year due to the absence of regrowth under the constraining climate of the site, two different shapes for pieces were used to distinguish between during spring or summer grazing. Rules for a given year defined constraints for grazing or mowing as how many pieces of each type could be allocated to each cell depending on land-management types and on grassland production for each land-management type according to the climate of the session. The game started with the board game filled with pieces allocated according to each farmer's managing system. Each of the four rounds then followed five steps: (1) reading of storylines by the game coordinator, (2) distribution of cards giving practical information on grassland production constraints expressed as number of pieces allowed by cell type, (3) during the action itself players may move pieces, remove or add pieces from the board and talk together, (4) summary of fodder and herd outcomes for each farmer on a purpose-designed table, (5) group debriefing about main actions. The first round was given a favourable climate year enabling farmers to understand the game and to move pieces around in case they disagreed with the initial state as compared with their own vision of their farm system.

Each game session ended with a debriefing session which allowed players to debate about the relationships between the potential actions taken during the game and the real world (Barreteau et al., 2007). The entire game then ended with a general discussion of how socio-economic alternatives may favour or constrain adaptation of each farmer in response to each of the climate alternatives. We presented the parameters of socio-economic alternatives to farmers based on graphs and landscape drawings, and their effects on adaptations to each climate alternative were discussed and summarized on a paper board. This final step allows us to tease out climate from socio-economic effects in our analysis of adaptation pathways. The full game, including debriefings, lasted five hours (2 hours the morning and 3 hours the afternoon), and was video recorded.

3.1.3 Validation of game adaptation and land management changes

Land use change models need to be carefully validated if they are to be used to support collective decision-making or action (Lambin et al., 2003). To strengthen the validation process, we cross-checked information obtained from different sources. First, on the day following the game the relevance of game actions was discussed individually with the farmers to place them out of the idealized space offered by the game. In addition these interviews allowed us to understand individual rationales underlying the actions during the RPG and to discuss the question of the difference between the RPG and reality (Castella et al., 2005). In addition to debriefing interviews, available narrative extracts from the RPG video were used to identify spatially-explicit biophysical or socio-economic drivers of land-use decisions (Washington-Ottombre et al., 2010). Second, proposed adaptations were assessed in a broader context with technicians from local agricultural and environmental extensions services so as to validate their feasibility.

3.1.4 Generation of land management scenario maps

The game board state was transcribed into a GIS (ArcGIS 10, ESRI) at the end of each round during both sessions. First practises applied in each cell were translated into land management types (see Table 1) based on a land-use state and transition model (Quétier et al., 2007b). Second we rescaled the board to the actual landscape by calculating the proportion between the number of cells of each land management type by each farmer on the board and real land parcels.

Moreover, to downscale the scenarios combining climate and socio-economic alternatives to a spatially-explicit format, we created rules based on statistical analysis of current land management types and biophysical constraints. Modifications or conversions of land management types were translated as a transition from one type to another (e.g. conversion from a mown unterraced grasslands to a grazed unterraced grasslands), except for the fertilization on grazed terraces and unterraced grasslands outside terraces which lead to new type of land management (Figure 2 and 3). The possibility of mechanisation of parcels due to slope was checked for mowing and fertilisation by statistical analysis of the spatial distribution of current practises. The currently mechanized parcels (manured and/or mown) have a 3rd quartile slope of 18° and an average of 15°, hence a slope below 18° degree was considered mechanisable and easily mechanisable at 15°. The maximum potential manured surfaces were calculated for each farm based on their herd size in each scenario with the help of technician from agricultural extension services (the reference for calculation was an average of 20 T/ha of manure spread every three years and a production of manure of 4.5 T/Livestock Unit). Other decision criteria such as distance to the farm, time of travel or difficulties to reach the parcels were not taken into account quantitatively in the simple model presented here due to the level of data required and because they were considered less restrictive than slope by farmers. In addition, any mention to a specific parcel by a farmer was taken into account.

4 Results

4.1 Climate and socio-economic scenarios

The two workshops with experts lead to four scenarios combining two socio-economic alternatives and two climate alternatives with relevant details for the study area (Table 2) in terms of general drivers and also on expected consequences on grasslands. The "*intermittent*" climate context expressed an increase in the frequency of spring or summer drought periods alternating with wetter years, according to the recent situation in some regions of the Alps. The "*drastic*" climate context included four consecutive years of springtime drought. Effects of drought on vegetation were considered very weak in the "*intermittent*" context consistent with the high resilience of the alpine vegetation against recent droughts and experimental drought simulations (Benot et al., in revision).

In the "*international*" socio-economic context, stakeholders faced continuing globalisation and urban concentration, with agriculture being supported only for its role as a producer of global environmental services (e.g. carbon storage, open landscapes). In the "*local*" context, as citizens showed a growing interest in their geographic area and its activities, and subsidies to mountain farming were maintained allowing it to remain a producer of quality food in line with strict environmental requirements (Table 2).

	Climate alternatives	
Drivers	**"Drastic"**	**"Intermittent"**
Temperature rise	No	No
Season of drought and occurrence	Spring drought during four consecutive years	Spring or summer drought every two years
Effects on vegetation	Change in species composition. Development of species adapted to drought (eg. *Festuca paniculata, Carex sempervirens*)	No change
Effects on biomass production	Decrease by more than 50%	Decrease by 15% during drought years
Effects on water quantity (springs)	Decreased flow of all springs, even quenching of the less productive ones	Decrease flow of the weaker springs
	Socio-economic alternatives	
	"Local"	**"International"**
Consumption demand	Local and high quality products	Cheapest prices
Aim of agricultural subsidies	To maintain both an agriculture with quality production and a high level of ecosystem services and biodiversity conservation. High subsidies but more restrictive in term of expected outcomes than in the "International" alternative.	To maintain open landscapes and production of environmental services such as carbon sequestration. Lower subsidies than on the local alternative, but less restrictive. A minimal income is guaranteed to farmers
Agricultural input prices (fodder, straw)	Variations due to climate	Variations due to climate
Agricultural product prices	15% decrease only for conventional products	15% decrease even for quality labelled product and – 10% for organic products
Part-time job opportunities	On-farm pluri-activity linked to agri-tourism	Off-farm pluri-activity job opportunities

Table 2: Scenarios drivers and related assumptions. Example of the corresponding storylines for the "drastic and local" scenario is presented as Supplementary Material (Appendix 1).

4.2 Projections of land management change

Despite the complexity of the multiple colours and pieces on the board game, farmers enter quickly into the game. While the board game is in part fictitious, farmers recognize which sector in the landscape each cell belongs to and then take care to reproduce realistic management during the RPG. The two RPG sessions lead to contrasted landscape configurations in term of land management (Figure 2). Two core land management changes were proposed by farmers, shift from mowing to grazing and increase in manuring. (Figure 3).

In the "*drastic*" climate alternative almost all farmers ceased mowing due to the strong decrease in grassland production and to the priority given to sustain grazing. Remaining mown surfaces made up 8% of the landscape in the "*drastic and local*" (only in unterraced grasslands) and 12 % in the "*drastic and international*" scenario (9% in terraces, 3 % in unterraced grasslands), compared with the current 28% (Figure 3). Farmers had to compensate fodder supply by purchases, regardless of economic difficulties. The "intermittent" alternative was seen as business-as-usual by farmers (Figure 3: lower part) with frequently recurring droughts over the last decades. The annual time step of the game allowed us to observe reduction in the extent of mowing were caused by summer rather than spring droughts. Farmers replenished fodder stocks during good years to buffer supply in drought years. Overall, under the "intermittent" alternative mowing was reduced to 27-18% of the landscape in the local and international socio-economic alternatives respectively.

Increase manuring was seen by farmers as a means of increasing grassland production and thereby fodder yields and stocking rates in both scenarios. The proportion of manured surfaces increased from 8 % (current) to 16 % under the "*local*" socio-economic alternative whether on grazed surfaces under the drastic alternative or mown surfaces under the "intermittent" alternative. Similarly, manured surfaces reached 15 % and 13% in the "*intermittent-international*" and "*drastic-international*" (almost exclusively mown surfaces) scenarios respectively.

Socio-economic alternative appeared to affect adaptation particularly regarding the decision to maintain mowing in less mechanisable parcels, and herd size. In the "*international*" alternative, less restrictive CAP subsidies on the minimum stoking rate and on required practises allowed farmers to decrease their herd and to restrict mowing to the more easily mechanizable parcels. In contrast, in the "*local*" alternative, herd size could be decreased thanks to the value added by direct sales. Moreover, the context of the "*international*" alternative was not favourable to new farmers taking over land of retired farmers, hence parcels were redistributed among current farmers (redistribution of 15 % area in the "drastic and international" scenario; 5% in the "intermittent and international" or the "drastic and local" scenario; and 1% in the "intermittent and local" scenario), allowing them to maintain or only slightly decrease their herd.

Beyond effects on the farm functioning in itself, these land management changes translated to strong landscape effects by modifying the proportion and location of different land management types. Figure 2 shows that changes occured mainly in the lower part (terraces), and in the *international* context also in mown unterraced grasslands. The *drastic and local* scenario incurred the most dramatic change with all the terraces managed only by grazing (about a doubling, with grazing on terraces increasing from 12 % to 27 % of the total area, of which 14% fertilized).

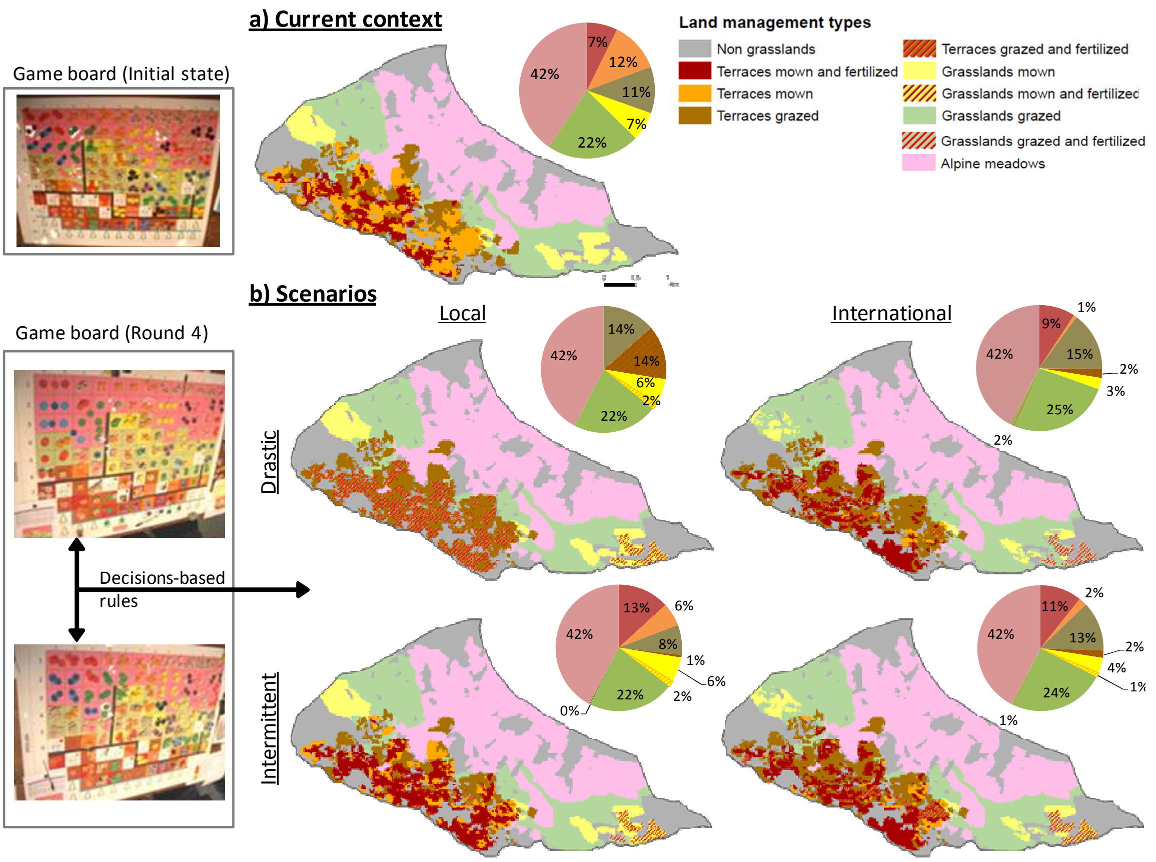


Figure 2: a. Current land management on the Villar d'Arène landscape; b. Scenarios maps obtained from the analyses of the board games and rule-based decisions on slope constraints to mechanization. Pie charts give the percentage of each land management type in the landscape.

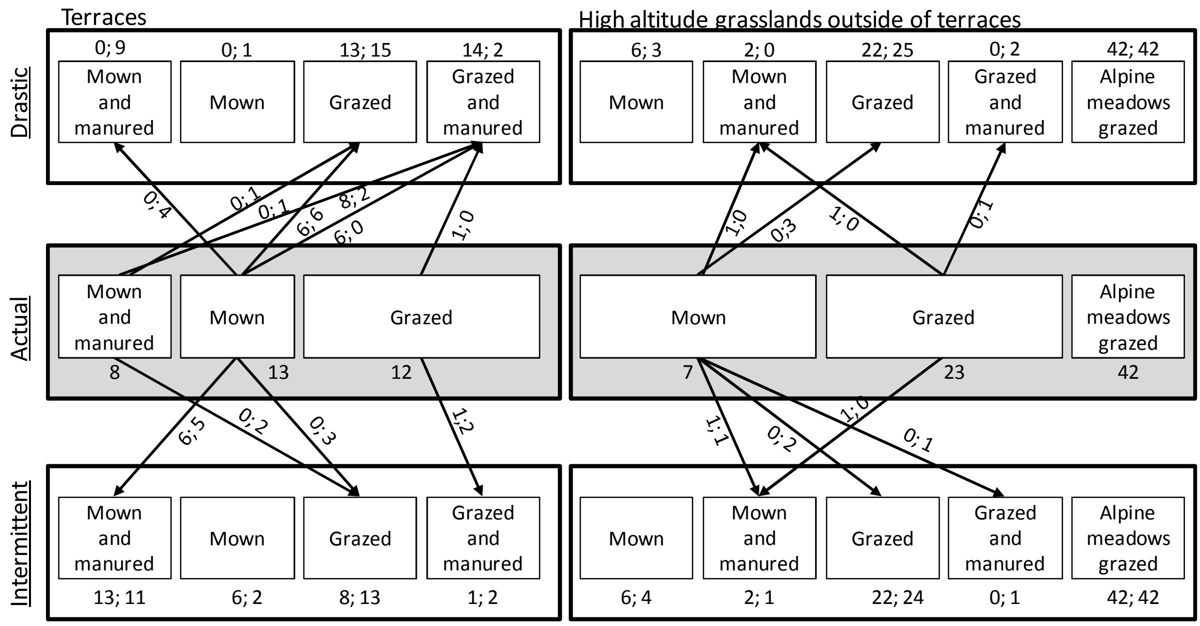


Figure 3: Land management transitions under the four scenarios. Climate alternatives are described by the upper and lower rows of boxes and numbers give the percentage (a;b) of land in the landscape respectively for: (a) the "local" alternative and (b) the "international" alternative. Arrows show transition from one land management type to another.

5 Discussion

The main objective of this study is to test a methodology for generating spatially-explicit projections of land management under different climate and socio-economic scenarios. We first discuss advantages and limits of the method. Then, we discuss projected land management changes in the case of Villar d'Arène and their potential impacts.

5.1 Methodological relevance and opportunities

Like previous studies using participative approaches (Alcamo et al., 2006; van Vliet et al., 2010), the proposed methodology meets to a great extent the four criteria for the evaluation of scenarios (Alcamo et al., 2006): relevance (How relevant are scenario for end users ?), credibility (Is the scenario plausible and consistent with existing information?), legitimacy (Does the scenario reflect points of view that are perceived to be fair?) and creativity (Do the scenarios challenge the current view of the future ?, Do they inform their audience about the implications of uncertainty?). Relevance and legitimacy are particularly important concerns because they strongly affect the stakeholders willingness to participate. Addressing meaningful issues for stakeholders in relation with reality is indeed very important to compensate for the time consuming aspect of the method (Walz et al., 2007).

Relevance, creativity and credibility are strengthened by the originality of the method constructing land-use scenarios in two steps, involving in step part those who know best what is appropriate, feasible and acceptable in their respective contexts: (1) climate and socio-economic scenarios development with experts and (2) effects of scenarios on land management with farmers by role playing. Highly detailed storylines and translation into the RPG increased credibility and legitimacy of climate and socio-economic scenarios because simulation of future contexts makes stakeholders understand easily which and how changes occur. Involvement of a diversity of regional experts was efficient to enhance scenario creativity (as in van Vliet et al., 2010; Volkery et al., 2008) by sharing a diversity of knowledge covering a variety of disciplines and sources, as well as technical and empirical as well as scientific knowledge, during the third first phases of scenarios development (Figure 1). Moreover, experts were for some of them potential users of scenarios and hence contributed to increasing scenario relevance. In the following we first examine how the approach met the six objectives proposed for the role playing game. We then briefly discuss the game implementation in other contexts.

The criteria for which a role playing game were chosen in comparison to other participatory methods (see section 2.3.2) appear to have been reached overall. Role-playing with a close to real landscape on the board game instead of interviews made it possible to translate qualitative storylines into quantitative and spatially land management projections (objective 2). Other techniques such as interviews based on a hand-drawn map or participatory land use mapping could have been used for this type of study but the role-playing game had the advantages of putting people in the future contexts and at the same time testing adaptation in a « concrete » situation, instead of only imagining adaptations out of context (objective 1). Moreover, unlike in group interviews RPG allow shy persons to propose adaptations as they can play without talking.

By combining a board game with a post-game discussion, we were able to model land management change both in terms of conversions from one management type to another (e.g. grazing replacing mowing) and as modifications which are more subtle changes (e.g. livestock stoking rate or fertilization) affecting the character of the land management within its category (Lambin et al., 2003). Moreover, focussing the game only on climate change and discussing afterwards in terms the effect of socio-economic changes on adaptations allowed us to separate the effects of climate alternatives alone from their combination with socio-economic alternatives.

Presenting scenarios to farmers with a yearly time step increased their understanding (objective 1) (as in Duru et al., 2012a). For example, while the "intermittent" alternative was quickly considered as business as usual by farmers, the "drastic" alternative, which was first considered more unlikely due to difficulties in imagining adaptations, was progressively seen as more plausible because some changes or adaptations proposed during the game reminded them of real past situations. Against our expectations, no group decisions were taken despite discussions among farmers about the opportunity of such collective decisions. On the other hand, expected neighbourhood effects occurred such as discussion between players, advices, and imitation between farmers, much like in real life (objective 4). Finally, the game afforded us a better understanding of the system which substantially improved our game model for future uses (objective 5 and 6) (as in Duru et al., 2012a). This role playing game was conceived as a model based on a set of assumptions to represent the system and aimed at answering scientific research questions rather than giving real technical advices. At the same time, the objective towards stakeholders was to raise awareness of possible future changes (Barreteau et al., 2010a), more specifically on farm system vulnerability and effect of adaptations on land-use changes (objective 6). Making sure to explain from the beginning who and for what purpose each farmer is invited to participate is important to prevent potential disappointment (Barreteau et al., 2010). The game process and its outcomes interested farmers and some said afterwards "*when I came back home, I thought to all these things we discussed during the RPG. It was formative!*". The farmers raised questions in the debriefing and/or interviews about technical advice. Moreover, this kind of process builds trust between participants and researchers (Castella et al., 2005). Farmers said at the end of the RPG "*this game is better than just presentation and discussion, we felt as actors of the study*" (objective 6). We suspected a learning effect from the morning to the afternoon session regarding anticipation of potential changes. After playing in the morning session without having expected such repeated drought (drastic alternative), farmers started to fertilize during the afternoon session (intermittent alternative) to increase forage production during good years so as to buffer a potential future drought. Inversion of RPG sessions would perhaps have not led to same results. This emphasizes the importance of methodological choices, because the game implementation is not neutral on results.

The small size of the study area allowed us to test the RPG with the totality of farmers. Nevertheless, like in other RPG (Barnaud et al., 2008) the framework may be applied in a larger area with a sample of farmers based on archetypes of farmers' behaviour such as those described by Gibon et al. (2010) for the Pyrenees. Indeed, this typology is to a large extent consistent with farmer behaviours in our study area and could thus be used for sampling given the similar agricultural contexts. Finally, although this RPG focused on drought impacts, the same RPG could be applied for any other change affecting forage resources such as recent vole outbreaks (Lucas, 2010; Quéré et al., 1999).

5.2 Adaptation strategies and land management changes in Villar d'Arène

The agricultural area of Villar d'Arène historically (around 1810) used at ca. two thirds % for cereal and potatoe production went through gradual changes until the total abandonment of ploughing on terraces in the mid or late 1970s (Girel et al., 2010). Since then, a strong conversion of land-use has taken place. In spite of these dramatic changes, when presented with scenarios farmers felt uneasy about adaptations going beyond simple land management modifications. Adaptations carried-out during the RPG were mostly tactical and reflect their experience in similar recent drought events, e.g. with the purchase of fodder to complement stocks (Nettier et al., 2010). Although some strategic adaptations proposed by some farmers during the fame are actually underway in the current economic context (e.g. decrease of livestock size by added value from direct sales). The presence of a large area of lightly stocked alpine meadows helped buffer inter-annual climate variations (Lemaire and Pflimlin, 2007) in the intermittent climate alternative, but not in the drastic climate alternative where even their production decreased. The observed adaptations were also influenced by socio-economic alternatives through difference in types of subsidies. For example in the local alternative agri-environmental measures provided high support to maintain biodiversity in unterraced grasslands, therefore farmers considered the opportunity cost of converting them to grazing.

Overall, farmers appeared quite conservative and limited change as much as possible. While in some areas the possibilities of off-farm activities or non-agricultural activities on farms enabled by tourism result in a decrease in herd size (Garcia-Martinez et al.), pluri-activity was in our case more a consequence of income loss following forced decrease in herd size as a result of lack of fodder, than a motivation in itself. But non-economic motives are also important in land-use change (Strijker, 2005). Therefore, adaptations lead mainly to modifications of land management and only to one type of conversion of land management (but not land-cover), from mowing to grazing, again more as a result of natural environmental constraints (topography, climate, short growing season, low productivity) induced by high mountain context (Mottet et al., 2006; von Glasenapp and Thornton, 2011) than by endogenous motivation. While some studies have shown that grazing represents an intermediate option between hay making and total abandonment (MacDonald et al., 2000; Mottet et al., 2006), abandonment was never observed during the game because Villar d'Arène's farmers consider they must keep meadows in good state (Quétier et al., 2010b).

Results showed that drought occurrence affects farming strategies and land management. However, consistent changes across scenarios regarding the specific location of changes pointed out the most vulnerable areas in the current landscape and farming system. For example, areas where mowing is ceased consistently across scenarios are more likely to be abandoned in the future. Changes such as conversions from mowing to grazing, or increased manuring can in the mid- to long-term affect grasslands floristic composition and functional traits properties (Niedrist et al., 2009; Quétier et al., 2007b; Rudmann-Maurer et al., 2008) and hence ecosystem services (Lavorel et al., 2011), but also cultural heritage such as through the erosion of terraces by trampling. Environmental policy should consider direct and indirect effects of climate change on the delivery of ecosystem services because they will remain essential for these agricultural systems for which environmental results are increasingly expected.

Acknowledgments

We thank all participants of workshops and the role playing game for their contributions and the time they devoted to our study. We also thank Fabien Quétier for his useful advices and help during the RPG session, and the Joseph Fourier Alpine Station for logistical support. This research was conducted on the long term research site "Zone Atelier Alpes", a member of the ILTER-Europe network (ZAA publication n°XXXXX). This work was supported by ANR through the FP6 BiodivERsA Eranet VITAL project (ANR-08-BDVA-008) and by the French Environment Ministry (MEEDDM) through the GICC-2 SECALP project.

References

Agrimonde®, 2009, Agricultures et alimentations du monde en 2050 : scénarios et défis pour un développement durable. Note de synthèse. INRA et Cirad (ed.), Paris.

Alcamo, J., Kok, K., Busch, G., Priess, J. A., Eickhout, B., Rounsevell, M., Rothman, D. S., Heistermann, M., Lambin, E. F., Geist, H., 2006, Searching for the Future of Land: Scenarios from the Local to Global Scale

Land-Use and Land-Cover Change, Springer Berlin Heidelberg, pp. 137-155.

Barnaud, C., Trebuil, G., Dumrongrojwatthana, P., Marie, J., 2008, Area study prior to companion modelling to integrate multiple interests in upper watershed management of Northern Thailand, *Southeast Asian Studies* **45:**559-585.

Barnaud, C., Van Paassen, A. M., Trébuil, G., Promburom, T., Bousquet, F., 2010, Dealing with power games in a companion modelling process: lessons from community water management in Thailand highlands, *Journal of International agricultural and extension education* **16**(1).

Barreteau, O., Bots, P. W. G., Daniell, K. A., 2010, A Framework for Clarifying "Participation" in Participatory Research to Prevent its Rejection for the Wrong Reasons, *Ecology and society* **15**(2).

Barreteau, O., Bousquet, F., Attonaty, J. M., 2001, Role-playing games for opening the black box of multi-agent systems: method and lessons of its application to Senegal River Valley irrigated systems, *Jasss-the Journal of Artificial Societies and Social Simulation* **4**(2)**:**U75-U93.

Barreteau, O., Le Page, C., Perez, P., 2007, Contribution of simulation and gaming to natural resource management issues: An introduction, *Simulation & Gaming* **38**(2)**:**185-194.

Benot, M. L., Saccone, P., Reydet, E., Vicente, R., Colace, M. P., Grigulis, K., Clément, J. C., Lavorel, S., in revision, Management, not summer climate manipulation, drives changes in biodiversity and functioning of a subalpine grassland, *Oecologia*.

Bigot, S., Rome, S., Biron, R., Laurent, J.-P., 2010, Mesures geophysiques (air-sol) a l'echelle d'une pelouse des hauts

plateaux du Vercors : analyse des variations hydroclimatiques locales et regionales (H. Q. *Colloque de l'Association Internationale de Climatologie* (ed O.P. Vincent Dubreuil, Valerie Bonnardot), ed.), Rennes, pp. pp. 77-82.

Boe, J., Terray, L., Habets, F., Martin, E., 2006, A simple statistical-dynamical downscaling scheme based on weather types and conditional resampling, *Journal of Geophysical Research-Atmospheres* **111**(D23).

Bohunovsky, L., Jager, J., Omann, I., 2010, Participatory scenario development for integrated sustainability assessment, *Regional Environmental Change* **11**(2)**:**271-284.

Borjeson, L., Hojer, M., Dreborg, K. H., Ekvall, T., Finnveden, G., 2006, Scenario types and techniques: Towards a user's guide, *Futures* **38**(7)**:**723-739.

Bossy, S., 1985, Associations foncières pastorales et groupements pastoraux : bilan d'une décennie *Revue de Géographie Alpine* **73**(4)**:**439-463.

Bryant, C. R., Smit, B., Brklacich, M., Johnston, T. R., Smithers, J., Chiotti, Q., Singh, B., 2000, Adaptation in Canadian agriculture to climatic variability and change, *Climatic Change* **45**(1)**:**181-201.

Carpenter, S. R., Pingali, P. L., Bennett, E. M., Zurek, M. B., 2005, Ecosystems and Human Well-being: Scenarios. Volume 2: the Millenium Ecosystem Assessment. Washington DC.

.

Castella, J. C., Trung, T. N., Boissau, S., 2005, Participatory simulation of land-use changes in the northern mountains of Vietnam: the combined use of an agent-based model, a role-playing game, and a geographic information system, *Ecology and society* **10**(1).

Coreau, A., 2009, Dialogue entre des chiffres et des lettres. Imaginer et construire des futurs possibles en écologie, Université Montpellier II, pp. 524.

de Jouvenel, H., 2002, La démarche prospective. Un bref guide méthodologique, *Futuribles* **247:**1-24.

Deboeuf, E., 2009, Adaptabilité des systèmes d'élevage de haute-montagne à des aléas. Le cas de Villar d'Arène, Enita de Clermont-Ferrand, France, pp. 91.

Duru, M., Felten, B., Theau, J., Martin, G., 2012, A modelling and participatory approach for enhancing learning about adaptation of grassland-based livestock systems to climate change, *Regional Environmental Change***:**1-12.

Engler, R., Randin, C. F., Thuiller, W., Dullinger, S., Zimmermann, N. E., AraÚJo, M. B., Pearman, P. B., Le Lay, G., Piedallu, C., Albert, C. H., Choler, P., Coldea, G., De Lamo, X., DirnbÖCk, T., GÉGout, J.-C., GÓMez-GarcÍA, D., Grytnes, J.-A., Heegaard, E., HØIstad, F., NoguÉS-Bravo, D., Normand, S., PuŞCaŞ, M., SebastiÀ, M.-T., Stanisci, A., Theurillat, J.-P., Trivedi, M. R., Vittoz, P., Guisan, A., 2011, 21st century climate change threatens mountain flora unequally across Europe, *Global Change Biology* **17:**2330-2341.

Etienne, M., 2003, SYLVOPAST: a multiple target role-playing game to assess negotiation processes in sylvopastoral management planning, *Jasss-the Journal of Artificial Societies and Social Simulation* **6**(2).

Garb, Y., Pulver, S., VanDeveer, S. D., 2008, Scenarios in society, society in scenarios: toward a social scientific analysis of storyline-driven environmental modeling, *Environmental Research Letters* **3**(4).

Garcia-Martinez, A., Bernues, A., Olaizola, A. M., Simulation of mountain cattle farming system changes under diverse agricultural policies and off-farm labour scenarios, *Livestock Science* **137**(1-3)**:**73-86.

Gibon, A., Sheeren, D., Monteil, C., Ladet, S., Balent, G., 2010, Modelling and simulating change in reforesting mountain landscapes using a social-ecological framework, *Landscape Ecology* **25**(2)**:**267-285.

Girel, J., Quétier, F., Bignon, A., Aubert, S., 2010, Histoire de l'agriculture en Oisans. Hautes Romanche et pays faranchin. Villar d'Arène, Hautes-Alpes, in: *La Galerie de l'Alpe*, Station Alpine Joseph Fourier, Grenoble, France, pp. 79.

Grard, M., 2010, Le rôle des politiques publiques dans les services écosystémiques des prairies de montagne, Master sciences et politiques de l'environnement, UMPC-IEP, Paris.

Jäger, J., Rothman, D., Anastasi, C., Kartha, S., van Notten, P., 2008, Training Module 6. Scenario development and analysis. GEO Resource Book. A training manual on integrated environmental assessment and reporting. Available at: http://www.unep.org/ieacp/iea/training/manual/module6.aspx.

Lambin, E. F., Geist, H. J., Lepers, E., 2003, Dynamics of land-use and land-cover change in tropical regions, *Annual Review of Environment and Resources* **28:**205-241.

Lavorel, S., Grigulis, K., Lamarque, P., Colace, M.-P., Garden, D., Girel, J., Pellet, G., Douzet, R., 2011, Using plant functional traits to understand the landscape distribution of multiple ecosystem services, *Journal of Ecology* **99**(1)**:**135-147.

Lavorel, S. c., 2011, Adaptation des territoires alpins à la recrudescence des sécheresses dans un contexte de changement global (SECALP). Rapport de fin de contrat. LECA CNRS-Université J. Fourier Grenoble, Cemagref Grenoble, Parc National des Ecrins Gap.

Lemaire, G., Pflimlin, A., 2007, Les sécheresses passées et à venir: quels impacts et quelles adaptations pour les sysèmes fourragers, *Fourrages* **191:**163-180.

Lucas, E., 2010, Analyse, bilan et perspectives de gestion des pullulations de campagnols terrestres dans le secteur du Briançonais, Université Joseph Fourier, Grenoble 1, Grenoble.

MacDonald, D., Crabtree, J. R., Wiesinger, G., Dax, T., Stamou, N., Fleury, P., Gutierrez Lazpita, J., Gibon, A., 2000, Agricultural abandonment in mountain areas of Europe: Environmental consequences and policy response, *Journal of Environmental Management* **59:**47-69.

Martin, G., Felten, B., Duru, M., 2011, Forage rummy: A game to support the participatory design of adapted livestock systems, *Environmental Modelling & Software* **26**(12)**:**1442-1453.

Metzger, M. J., Rounsevell, M. D. A., Van den Heiligenberg, H., Perez-Soba, M., Hardiman, P. S., 2010, How Personal Judgment Influences Scenario Development: an Example for Future Rural Development in Europe, *Ecology and society* **15**(2).

Millennium Ecosystem Assessment, 2005, Ecosystems and human well-being: scenarios, Island Press, Washington D.C., USA.

Mora, O. c., 2008, Les nouvelles ruralités à l'horizon 2030. Des relations villes campagnes en émergence ?, Quae ed., Paris.

Moss, R. H., Edmonds, J. A., Hibbard, K. A., Manning, M. R., Rose, S. K., van Vuuren, D. P., Carter, T. R., Emori, S., Kainuma, M., Kram, T., Meehl, G. A., Mitchell, J. F. B., Nakicenovic, N., Riahi, K., Smith, S. J., Stouffer, R. J., Thomson, A. M., Weyant, J. P., Wilbanks, T. J., 2010, The next generation of scenarios for climate change research and assessment, *Nature* **463**(7282)**:**747-756.

Mottet, A., Ladet, S., Coque, N., Gibon, A., 2006, Agricultural land-use change and its drivers in mountain landscapes: A case study in the Pyrenees, *Agriculture Ecosystems & Environment* **114**(2-4)**:**296-310.

Naivinit, W., 2009, Modélisation d'accompagnement pour l'analyse des interactions entre usages des terres et de l'eau et migrations dans le bassin versant de la Lam Dome Yai au Nord Est de la Thaïlande, in: *Géographie humaine, économique et régionale*, Université Paris Ouest Nanterre-La Défense, pp. 344.

Nettier, B., Dobremez, L., Coussy, J. L., Romagny, T., 2010, Attitudes of livestock farmers and sensitivity of livestock farming systems to drought conditions in the French Alps, *Revue De Geographie Alpine-Journal of Alpine Research* **98**(1)**:**383-400.

Niedrist, G., Tasser, E., Luth, C., Dalla Via, J., Tappeiner, U., 2009, Plant diversity declines with recent land use changes in European Alps, *Plant Ecology* **202**(2)**:**195-210.

Pachauri, R. K., 2007, Climate Change 2007: Synthesis Report. Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, IPCC.

Pagé, C., Terray, L., Boé, J., 2008, Projections climatiques à échelle fine sur la France pour le 21ème siècle : les scénarii SCRATCH08, in: *Technical Report TR/CMGC/08/64, Centre Européen de Recherche et de Formation Avancée en Calcul Scientifique (CERFACS)*.

Pak, M. V., Brieva, D. C., 2010, Designing and implementing a Role-Playing Game: A tool to explain factors, decision making and landscape transformation, *Environmental Modelling & Software* **25**(11)**:**1322-1333.

Quéré, J. P., Garel, J. P., Rous, C., B., P., P., D., 1999, Estimer les dégâts du Campagnol terrestre en prairie naturelle, *Fourrages* **158:**133_147.

Quétier, F., Rivoal, F., Marty, P., De Chazal, J., Lavorel, S., 2010a, Social representations of an alpine grassland landscape and socio-political discourses on rural development, *Regional Environmental Change* **10:**119-130.

Quétier, F., Rivoal, F., Marty, P., de Chazal, J., Thuiller, W., Lavorel, S., 2010b, Social representations of an alpine grassland landscape and socio-political discourses on rural development, *Regional Environmental Change* **10**(2)**:**119-130.

Quétier, F., Thebault, A., Lavorel, S., 2007, Plant traits in a state and transition framework as markers of ecosystem response to land-use change, *Ecological Monographs* **77**(1)**:**33-52.

Rouan, M., Kerbiriou, C., Levrel, H., Etienne, M., 2010, A co-modelling process of social and natural dynamics on the isle of Ouessant: Sheep, turf and bikes, *Environmental Modelling & Software* **25**(11)**:**1399-1412.

Rounsevell, M. D. A., Metzger, M. J., 2010, Developing qualitative scenario storylines for environmental change assessment, *Wiley Interdisciplinary Reviews-Climate Change* **1**(4)**:**606-619.

Rudmann-Maurer, K., Weyand, A., Fischer, M., Stocklin, J., 2008, The role of landuse and natural determinants for grassland vegetation composition in the Swiss Alps, *Basic and Applied Ecology* **9**(5)**:**494-503.

Schroter, D., Cramer, W., Leemans, R., Prentice, I. C., Araujo, M. B., Arnell, N. W., Bondeau, A., Bugmann, H., Carter, T. R., Gracia, C. A., de la Vega-Leinert, A. C., Erhard, M., Ewert, F., Glendining, M., House, J. I., Kankaanpaa, S., Klein, R. J. T., Lavorel, S., Lindner, M., Metzger, M. J., Meyer, J., Mitchell, T. D., Reginster, I., Rounsevell, M., Sabate, S., Sitch, S., Smith, B., Smith, J., Smith, P., Sykes, M. T., Thonicke, K., Thuiller, W., Tuck, G., Zaehle, S., Zierl, B., 2005, Ecosystem service supply and vulnerability to global change in Europe, *Science* **310**(5752)**:**1333-1337.

Strijker, D., 2005, Marginal lands in Europe - causes of decline, *Basic and Applied Ecology* **6**(2)**:**99-106.

Swetnam, R. D., Fisher, B., Mbilinyi, B. P., Munishi, P. K. T., Willcock, S., Ricketts, T., Mwakalila, S., Balmford, A., Burgess, N. D., Marshall, A. R., Lewis, S. L., 2011, Mapping socio-economic scenarios of land cover change: A GIS method to enable ecosystem service modelling, *Journal of Environmental Management* **92**(3)**:**563-574.

van Vliet, M., Kok, K., Veldkamp, T., 2010, Linking stakeholders and modellers in scenario studies: The use of Fuzzy Cognitive Maps as a communication and learning tool, *Futures* **42**(1)**:**1-14.

Vittoz, P., Randin, C., Dutoit, A., Bonnet, F., Hegg, O., 2009, Low impact of Climate change on subalpine grasslands in the Swiss Nothern Alps, *Global Change Biology* **15**(209-220).

Voinov, A., Bousquet, F., 2010, Modelling with stakeholders, *Environmental Modelling & Software* **25**(11)**:**1268-1281.

Volkery, A., Ribeiro, T., Henrichs, T., Hoogeveen, Y., 2008, Your Vision or My Model? Lessons from Participatory Land Use Scenario Development on a European Scale, *Systemic Practice and Action Research* **21**(6)**:**459-477.

von Glasenapp, M., Thornton, T., 2011, Traditional Ecological Knowledge of Swiss Alpine Farmers and their Resilience to Socioecological Change, *Human Ecology* **39**(6)**:**769-781.

Walz, A., Lardelli, C., Behrendt, H., Grêt-Regamey, A., Lundström, C., Kytzia, S., Bebi, P., 2007, Participatory scenario analysis for integrated regional modelling, *Landscape and Urban Planning* **81**(1-2)**:**114-131.

Washington-Ottombre, C., Pijanowski, B., Campbell, D., Olson, J., Maitima, J., Musili, A., Kibaki, T., Kaburu, H., Hayombe, P., Owango, E., Irigia, B., Gichere, S., Mwangi, A., 2010, Using a role-playing game to inform the development of land-use models for the study of a complex socio-ecological system, *Agricultural Systems* **103**(3)**:**117-126.

Supplementary materials

Storylines for the Drastic & local scenario

Dry years with intermittent wet (normal) years are observed. The first year, the weather is normal with wet winter and spring. But next year, spring is very dry (as in 2003 in France). Only 50% or 75% of usual yields are obtained in mown grasslands and when it's time to go on alpine meadows there is half the quantity of grasses(fodder?) and with an advance state of vegetation (already in flowering stage). Already weak springs run dry quickly and the others dwindle during the summer season. Summer storms occuring from mid-July allow regrowth of vegetation in alpine meadows as well as aftermath in grasslands. During the third year no drought occurs and forage is good in quality and quantity. But the fourth year, after a wet spring, a drought begins during summer (like in 2009 in France). Therefore, grass growth in quantity in mown grasslands (but quickly dry), but aftermath are not good to be mown. In alpine meadows, agronomic situation start to be difficult from mid-august. Usually, woody grasslands show better conditions of grass production than the other during drought. Climate of the following years goes on with the same alternation.

Along with this occurrence of drought, a socio-economic shift appears. Urban sprawl continues. Nevertheless, rural areas and society can benefit from an increasing demand on local product which reflects an image of quality and territorial anchorage (appealing?). This maintains the prices of direct selling product and/or organic or labeled (PDO or PGI European label), while prices of conventional products are falling. What society also likes about rural areas is doing leisure based on natural and cultural heritage, which promotes agri-tourism. Following the new CAP reform, farmers stop to benefit from first pillar subsidies. But, in a context of increasing requirements relative to environmental impact of agriculture, any new aid is now coupled with environmental criteria (Agri-environment measures) and provides payments to farmers who subscribe, on a voluntary basis, to environmental commitments related with obligation of results. Local administration and association promote rural development and contribute financially to new infrastructures or equipments necessary for agricultural and pastoral activities.

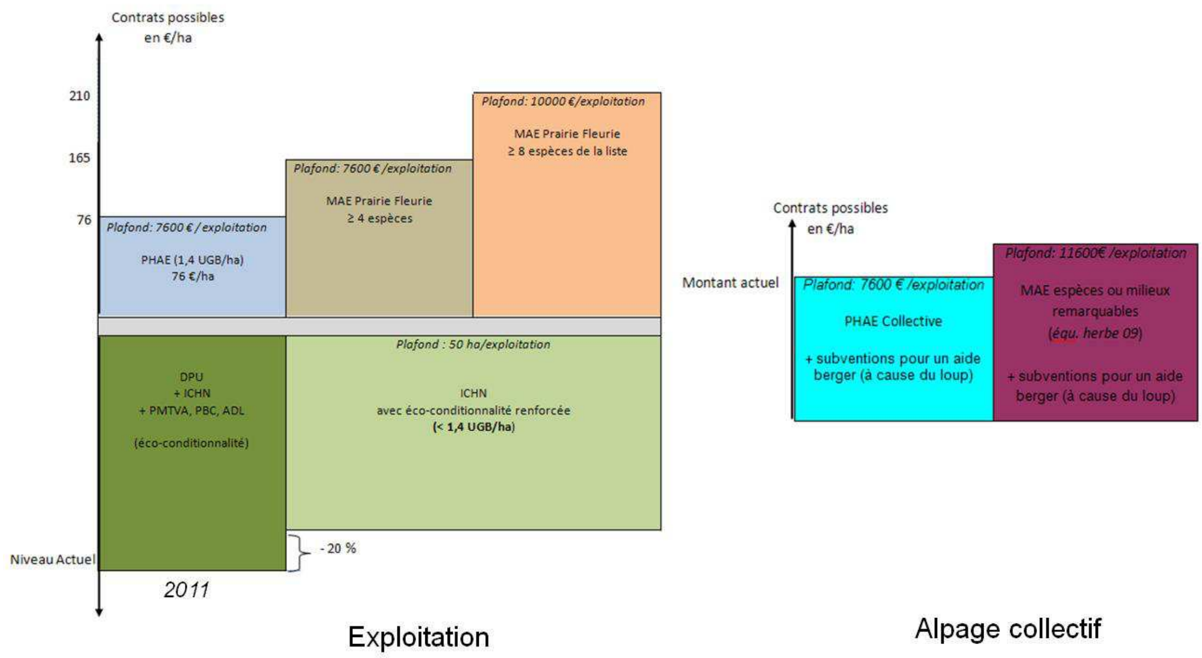


Figure S1 : Illustration showed to farmers about changes in agricultural policies (amount and types of subsidies) for the drastic and local scenario

Chapitre 6

Effets direct et indirect du climat sur les services des écosystèmes[12]

Abstract

Land-use and climate change are pointed out as the primary causes of global biodiversity loss and ecosystem services. However, while the consequences of climate change on ecosystem properties and associated services are well documented, unravelling the cascading impacts of climate change on ecosystem services through changes in agricultural management is largely overlooked. Here, we present a trait-based framework to understand how climate change is affecting trade-offs among ecosystem services under varying management conditions. Using alternative scenarios we discriminated direct effects of climate change on ecosystem functioning related to plant functional properties, from indirect effects through farmers' management adaptations. Ecosystem service supply was overall more sensitive to climate than to induced management change, but bundles ecosystem services remained stable across scenarios. However these responses were strongly influenced by the spatial extent of management change, as plot level effects on ecosystem properties need to be scaled up to the entire landscape. The trait-based approach revealed how interactions and trade-offs among ecosystem were determined by the combination of common driving traits and common responses to changes in fertility.

1 Introduction

The exponential growth in human activity is recognized as the main driver of Earth's environmental change (Steffen et al., 2007). Land-use and climate change are pointed out as the

[12] Ce chapitre fait l'objet d'un article en cours de préparation pour PNAS : Lamarque P*, Lavorel S*, Quétier F, Mouchet M, Direct and indirect effects of climate change on bundles of grassland ecosystem services. * Equal contribution. Dans cette étude j'ai réalisé l'ensemble des analyses statistiques et participé conjointement avec Sandra à l'élaboration du cadre conceptuel, de l'analyse des données et de la rédaction.

primary causes of global biodiversity loss (Carpenter et al., 2006; Sala et al., 2000), and hence strongly regulates the ability of ecosystem to provide the services required by human societies (Diaz et al., 2006; MEA, 2005). Mountain grasslands are increasingly considered for their multiple ecosystem services such as biodiversity, water quality, aesthetic value, fodder quality and quantity (Gibon, 2005b). However, while vulnerable mountain ecosystems are known as being particularly exposed to combined effects of climate and management change (Tappeiner and Bayfield, 2004; Gellrich and Zimmermann, 2007), studies untangling mechanisms underpinning the combined impacts of these two major drivers on ecosystem services are missing (Pereira et al., 2010). A key challenge to manage ecosystem services sustainably is to understand the interactions among multiple ecosystem services and in relation to drivers of change (Bennett et al., 2009; Carpenter et al., 2009; Millennium Ecosystem Assessment, 2005; Raudsepp-Hearne et al., 2010; Rodriguez et al., 2006). An increasing number of studies have analyzed multiple ecosystem services either as bundles defined as sets of services that appear together repeatedly (Raudsepp-Hearne et al., 2010), or identifying trade-offs (resp. synergies) defined by opposite (resp. parallel) responses of services to change (Bennett et al., 2009; Rodriguez et al., 2006). Although there is a clear need to unravel mechanisms leading to trade-offs among ecosystem services (Raudsepp-Hearne et al., 2010), studies addressing them are still rare. Bennett et al, (2009) demonstrated that trade-offs or synergies between ecosystem services can be based on two types of mechanisms: either due to direct interactions between ecosystem services, or due to effects of a common driver of change. The understanding of ES bundles requires a mechanistic approach rather than the widespread descriptive approach based on the exploration of the spatial co-occurrence of targeted services (e.g. (Chan et al., 2006; Nelson et al., 2009; Reyers et al., 2009) or between such services and biodiversity (e.g. (Anderson et al., 2009; Egoh et al., 2009)). Several methods have been proposed to identify trade-offs and synergies: pairwise correlation analyses (Chan et al., 2006), visualization of their simultaneously continuous variations with spider graphs (also called flowers or star diagrams, (Foley et al., 2005; Raudsepp-Hearne et al., 2010; Reyers et al., 2009; Rodriguez et al., 2006)) or the assessment of the strength of their relationships using multivariate analysis such as principal component analysis (Lavorel et al., 2011; Raudsepp-Hearne et al., 2010). However such methods are not designed to directly address the causes of associations among ecosystem services.

We addressed this fundamental knowledge gap by analysing mechanisms influencing ecosystem services and their relationships under different climate and land-use contexts. For this purpose, we used a scenario approach (chapter 5) which simulated the impacts of combined climate and land-use changes on ecosystem services using semi-mechanistic models of ecosystem properties based plant and microbial functional traits (Grigulis et al., submitted; Lavorel et al., 2011). Thereby, we were able to approach ecological mechanisms underpinning ES bundles and trade-offs, and how these may change in response to alternative scenarios.

Ecosystem service supply has been related to ecosystem biological characteristics (Kremen 2005), and more specifically to functional traits (de Bello et al., 2010; Kremen, 2005; Reiss et al., 2009). In particular, for plants there is growing evidence for the effects of community-level functional traits on ecosystem processes that underlie important ecosystem services (Diaz et al., 2007; Suding and Goldstein, 2008). Discovering that traits determining response to abiotic and land uses changes (e.g. fertilisation favours plants with nitrogen-rich leaves) are equal or correlated to traits that determine effects on ecosystem functioning (e.g. nitrogen-rich leaves at the community level promote high productivity of biomass) (Lavorel and Garnier, 2002) has markedly advanced the understanding of ecological constraints to and opportunities for the delivery of multiple ecosystem services (Hooper et al., 2005; Lavorel and Grigulis, 2012; Lavorel et al., 2011). The trait-based approach also makes it possible to analyse the effects on ecosystem services of detailed management practices, e.g. fertilisation effects (Duru et al., 2005; Duru et al., 2012b; Lavorel et al., 2011), as proposed by (Bennett et al., 2009)..

Considering that landscape scale changes in ecosystem services are the combined outcome of changes in management pattern, and of plot-scale effects of changed climate and management (Figure 1), we addressed three questions:

1) What are the effects of climate change directly or indirectly through land management adaptation on the supply of individual ecosystem services and on their bundles?
2) What are the relative contributions of direct and indirect climate effects on individual ecosystem services and their bundles?
3) Which are the mechanisms determining direct and indirect climate effects on ecosystem services bundles?

Using a grassland-dominated landscape from the French Alps as a case study, we quantified ecosystem service delivery based on a set of underlying measurable ecosystem properties (Lamarque et al., 2011a) (Chapter 4), that were identified by stakeholders as contributing to each ecosystem service (Haines-Young and Potschin, 2010). We hypothesised that trade-offs among ecosystem services result from functional mechanisms determining climate and land use effects on biodiversity and ecosystem functioning at plot scale, and from integrative effects at landscape scale resulting from changes in the representation of different land management types.

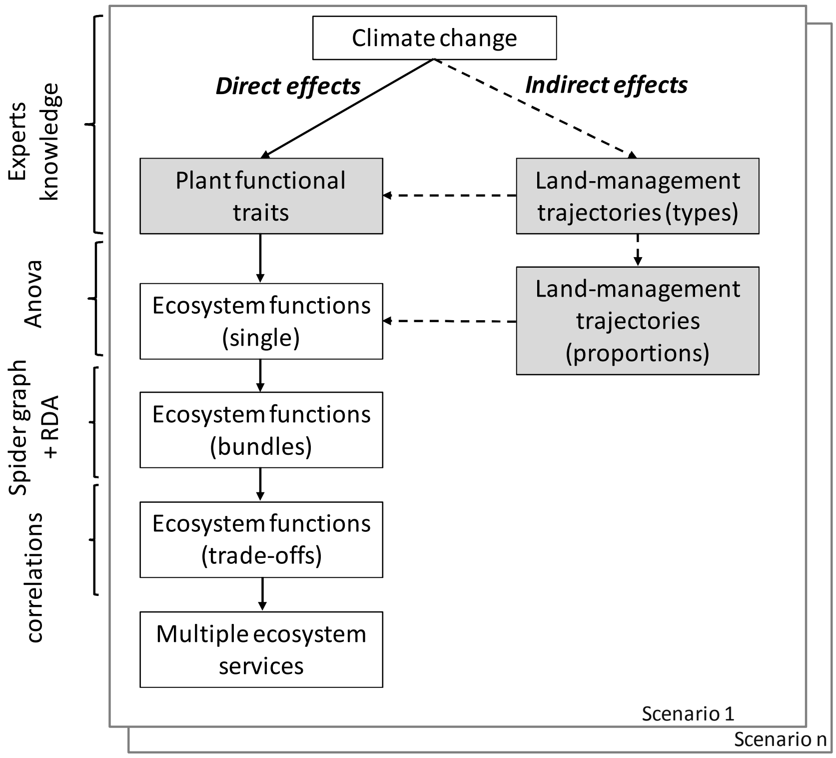


Figure 1: Conceptual framework for analysing effects of climate change on ecosystem services. The framework distinguishes a direct pathway of climate change effects on plant functional traits (plain arrows) from an indirect pathway through land-management adaptation resulting in change in land management types and/or proportions in the landscape(dotted arrows). Grey boxes indicate mechanisms underlying ecosystem services delivery. Quantitative methods used at each step are cited on the left side of the figure.

2 Study site and methods

2.1 Study site

The Lautaret site (45°03’ N, 6°24’ E) is located in the Central French Alps on the south-facing slopes of Villar d’Arène. The total area is 13 km² and the site ranges from 1552 to 2442 m a.s.l. It is used by intensive sheep and cattle livestock farming. Land management types, the combinations between past and present land management (Quétier et al., 2007b), describe the three main types of grasslands present in the site. Previously cultivated terraces (1550-1950m) are now manured and mown (LMT1), mown but not manured (LMT2) or unmown and grazed in spring and autumn (LMT3). Never cultivated unterraced grasslands (1700-2000m) with a multi-century history of mowing are currently mown (LMT4) or summer grazed (LMT5). Never mown summer grasslands (> 2200 m) are used since several centuries as private or collective alpine meadows (LMT7). Climate is subalpine with a strong continental influence due to a rain shadow with respect to dominant westerly winds. Mean annual rainfall is 956 mm and the mean monthly temperatures of -4.6°C in January to 11°C in July (at 2050 m above sea level). Rainfall occurs mainly during the cooler months, with 40% of annual rainfall during the growing season (April-September). But, since the last decades several droughts have occurred entailing a reduction up to 50% of forage production. These change lead farmers to adapt their practices such as purchasing fodder to compensate for the decrease in the harvests that normally provide herd feed in the winter leading to a conversion from mowing to grazing (Nettier et al., 2010) or increase fertilization to obtain more fodder (Lamarque et al,). A detailed site description can be found in Quétier et al.(2007b).

2.2 Combined scenarios of climate and land-use change

Four coupled climate/socio-economic regional scenarios simulating the evolution of the study site by 2030 were developed using a participatory approach taking into account regional experts' knowledge of future possible trends. These scenarios were then down-scaled to farm scale resulting in four land-management scenarios in collaboration with local farmers (Lamarque et al., submitted). The two climate alternatives expressed the consequences of alternative drought frequencies. In the "drastic" alternative droughts occur every years during spring, in any period of four consecutive years, leading to a decrease of forage production by up to 50% in comparison to wetter years. In contrast in the "intermittent" alternative, droughts occur every other year alternatively during spring or summer, leading to a decrease of forage production by 15% during drought years. The two socio-economic alternatives set the economical and political contexts which are a driving force behind much of the farmers' adaptation to climate change (Nettier et al., 2010). The "international" alternative sets an economical context based on globalisation of markets with decreasing prices of agricultural products except organic products, and reflects a political will to maintain agricultural ecosystem services production to meet global requirements, such as for carbon storage or water quality using non-binding subsidies. In the "local" alternative more concern about local product consumption and local development uphold prices of agricultural products. At the same time, policy supports are in favour of more environmentally friendly agriculture, requiring for example results in terms of species diversity. These scenarios led farmers to adapt their practises by increasing fertilisation (manure) to face decreasing forage production, or by favouring grazing at the expense of mowing to retain their herds in spite of lower forage yields, and buying fodder if farms economy allowed it (see Figure 2 stacked bar plots for percentage of change). For both climate alternatives, two alternative land management scenarios were thus obtained. Under the drastic climate alternative, mowing area decreased severely from 28% of the total area in the current landscape to 8% in the "Drastic-Local" and to 12% in the "Drastic-International" scenarios respectively. The fertilised area increased from 8% of the current landscape to 16% in the "Drastic-Local" and 13% in the "Drastic-International" either in grazed or mown area. Under the intermittent climate alternative, considered more as business as usual by farmers, the proportion of mown area decreased only slightly as compared to the current situation (28%), remaining at 27% in the “Intermittent-Local” scenario and decreasing to 18% in the “Intermittent-International” scenario. The fertilised area increased significantly under both scenarios, with 16% of landscape manured in the “Intermittent-Local” scenario and 15% in the “Intermittent-International” scenario.

2.3 Vegetation parameters and plant traits

The taxonomic composition of plant communities, plant traits (vegetative height, leaf dry matter content – LDMC, leaf nitrogen and phosphorus concentrations – LNC and LPC), environmental parameters (altitude, slope, water holding capacity – WHC, Nitrogen and phosphorus nutrition indices – NNI and PNI) were collected between 2003 and 2011 for 60 plots stratified by land management type, landscape sector, and altitude (for more details on data collection and measurements see Lavorel et al, 2011). The taxonomic diversity of plant communities was quantified using the Simpson’s index at plot level (i.e. alpha diversity) while plant functional diversity was estimated using community-weighted-mean (CWM, (Garnier et al., 2004)) and functional divergence (FD, Mason, 2003) of each functional trait separately.

Responses of these parameters to climate and land management change in each scenario were quantified by expert decision using a state-and-transition model (Quétier et al., 2007a) and results from field and pot experiments at the Lautaret site (Benot et al., in revision; Grassein, 2009), and in functionally close communities (Ansquer, 2006). Plant available nutrients parameters NNI and PNI decreased by 50% under drastic drought and 10% under intermittent drought due to limitation of nutrient availability to plant growth under water limitation. NNI and PNI increased in response to fertilization, with an increase twice greater under the “intermittent” alternative as compared to the “drastic” alternative so as to reflect limitation in nutrient availability as a result of soil water limitation (Table S1). Given the short time-frame of scenarios (2030) as compared to the slow dynamics of mountain grasslands which is constrained by low temperature and a short vegetative season, changes in floristic composition were described only in terms of changes in abundances, with no species turnover. Drought did not modify species abundances (Benot et al., in revision), while organic fertilisation of currently non-fertilised grasslands was assumed to result in a 3 fold increase in dicot abundances and a 30% increase in legume abundances. In addition, in unterraced grassland we assumed a shift among grasses with a 50% decrease in *Festuca nigra* and *Festuca paniculata* to the benefit of *Bromus erectus* (Quétier et al., 2007b). Plant functional traits were modified in the following way: height, LNC and LPC decreased and LDMC increased under the “drastic” alternative to reflect direct drought impacts on these traits at species level as assessed through experiments (Grassein, 2009) (Table S1). Under the “intermittent” alternative we assumed that direct drought effects on plant traits were negligible over the time scale of interest (Benot et al., in revision), and that they only responded to decreasing nitrogen fertility following statistical models by Lavorel et al. (2011). Projected Community Weighted Mean vegetative (CWM) were then calculated by combining changed composition and species trait values for each management type. Following Lavorel et al. (2011), Simpson diversity at the level of the species pool (i.e. gamma diversity) of each land management type was modified according to changes in NNI, either for a given management type between climate scenarios, or for newly fertilized management types according to the increase in NNI resulting from fertilisation. Assuming that beta diversity across plots remained constant, this allowed us to project alpha diversity values per plot under each scenario. Finally, based on multi-annual observations (Lavorel et al. unpublished), the flowering date of grasses advanced by about 21 days in the “drastic” alternative and 7 days in the “intermittent” alternative

2.4 Ecosystem properties

Variations across the landscape in ecosystem properties underpinning ecosystem services, green biomass production, litter mass, fodder crude protein content, plant diversity, date of flowering onset for grasses, nitrate retention, soil organic matter content, nitrogen mineralization potential, were modelled across the entire landscape for each 20 x 20 m pixel of land management maps applying general linear models (GLM) based on relevant plant and associated microbial traits and abiotic variables following Lavorel et al (2011) and (Grigulis et al., submitted) (Table S2).

In order to separate climate effects and land management effects, we designed a simulation experiment by creating, in addition to the current context and the four scenario combinations of climate and land management change, six additional artificial scenarios representing either land management scenarios with climate status quo (4 artificial scenarios) or climate scenarios with land management status quo (2 artificial scenarios). Statistical models of ecosystem properties were applied for these eleven climate and land use combinations including current conditions, the four actual scenarios and the six artificial scenarios (Table 1).

		Land management effects	
Climate effects		**NO**	**YES**
	NO	Current context : Status quo for climate and land management **LuCur-ClimCur** : Current Land use and climate context	4 artificial scenarios coupling land management scenarios' configurations with climate status quo **LuDI_ClimCur** : drastic and international land use with current climate **LuDL-ClimCur**: drastic and local land use with current climate **LuII-ClimCur**: intermittent and international land use with current climate **LuIL-ClimCur**: intermittent and local land use with current climate
	YES	4 artificials scenarios couling climate scenarios' effects with status quo land management configuration **LuCur-ClimD** : Current Land use and drastic climate **LuCur-ClimI**: Current Land use and intermittent climate	4 land-management scenarios coupled with climate scenarios' effects **LuDI-ClimD** : drastic and international **LuDL-ClimD**: drastic and local **LuII-ClimI** : intermittent and international **LuIL-ClimI**: intermittent and local

Table 1: Description of climate, land management and artificial scenarios used to analyse the relative contributions of direct and indirect effects of climate on ecosystem services and their bundles. The four land management scenarios were developed jointly with farmers (Lamarque et al, submitted), while the artificial scenarios were developed for the purpose of this study to observed change in land-use and climate separately.

2.5 Data analysis

Each of the 20 x 20 m pixel composing the landscape grid was characterized by an ecosystem property value (continuous) under each scenario, a scenario type coded using two categorical variables corresponding respectively to the land management and the climate alternatives (with 5 and 3 levels respectively; Table 1), a land management type (categorical variable with 8 levels), and mowing and fertilization described as binary variables. Data were aggregated for further analyses by summing pixel values across the entire landscape, or alternatively for each of the grassland types (terraced, unterraced, alpine meadows). Values for ecosystem properties were centred and scaled using the “scalewt” function (ade4 package) in order to have comparable values across properties. Analyses proceeded in three steps in order to address the three research questions.

First, we visualized changes in ecosystem properties and their bundles across scenarios for the entire landscape using spider graphs. This provided both a static view of synergistic and antagonistic ecosystem properties for a given scenario (Raudsepp Hearn et al. 2010), and a first dynamic view of trade-offs in response to scenarios (Rodriguez et al. 2009). Then, to further quantify interactions (synergies and antagonisms) between ecosystem properties, we estimated their pairwise Pearson's coefficients of correlation for each scenario (Chan et al., 2006).

Second, to estimate the relative contribution of 'climate' and 'land management' explanatory variables to the variation of individual ecosystem properties we used analysis of variance (ANOVA) followed by variance partitioning (see Table 2 for further details). P*ost hoc* Tukey HSD tests detected significant differences among treatments for explanatory variables which explained a significant amount of variation in a given ecosystem property. Then, to quantify and visualize the relative contributions of direct climate effects and of indirect effects via land management adaptation on trade-offs among ecosystem properties across scenarios we used a redundancy analysis (RDA) with climate' and 'land management' as explanatory variables for the matrix describing the eight ecosystem properties across the eleven scenario combinations. A second RDA was repeated replacing 'land management' by the explicit percentages of mown or fertilized pixels in combination with climate as explanatory variables.

Third, in order to identify specific mechanisms associated with scenario effects on landscape patterns, each statistical model was run for the entire landscape (sum of pixel values of the whole landscape) and also for individual grasslands types (sum of pixel values for terraces, non-terraces and alpine meadows respectively). As an additional aid for interpretation, effects of individual land management types were visualized by spider graphs describing ecosystem properties for each land management type within each scenario.

All statistical analyses were carried out with the R statistical software using the ade4 and vegan packages (R Development Core Team, 2008).

3 Results

3.1 Climate change effects on individual ecosystem properties and their bundles

Variations in individual ecosystem properties for the entire landscape were mostly driven by the direct climate effect pathway with strong differences in ecosystem properties between each alternative and the current climate (Figure 2, bottom left spider graph). Nitrogen mineralization, soil organic matter and nitrate retention increased under drastic drought while all other ecosystem properties decreased leading to a trade-off in responses between these two sets of ecosystem properties. Intermittent drought only decreased plant diversity and crude protein content, and brought on earlier grass flowering onset.

Ecosystem properties were much less responsive to alternative land management scenarios under status-quo climate (Figure 2, top right spider graph), with limited differences across scenarios. Only plant diversity and crude protein content, and to a smaller extent biomass production, were responsive to land management scenarios, increasing under the 'international local' scenario and decreasing under the 'drastic', and especially the 'drastic local' scenario. Combined effects of climate and land management scenarios (Figure 2, bottom right spider graph) were dominated by climate change effects (graphs mostly similar to climate only – bottom left). Additional land use effects regarded an enhanced loss in plant diversity under the 'drastic local' as compared to the 'drastic international' scenario, and a smaller increase in CPC accompanied by a smaller decrease in litter as compared to current conditions under the 'intermittent international' as compared to the 'intermitted local' scenario.

Patterns of correlation among ecosystem properties varied little across scenarios (Table S3). Of the 28 possible pairs of ecosystem properties, 11 pairs were highly correlated (Pearson coefficient; $r \geq 0.5$), of which 7 were synergies that were stable across scenarios (positive correlations: Litter-Gbio, CPC-SOM, PlantDiv-SOM, PlantDiv-CPC, NMP-SOM, NMP-PlantDiv). Under current management and climate, and similarly under land management change alone or intermittent drought, SOM, N mineralization, nitrate retention and CPC were compromised by biomass production, litter accumulation, plant diversity, late flowering onset of grasses (Figure 2), though only 2 negative correlations were strong (CPC-Litter, plantDiv-Litter). Drastic drought reversed this pattern by favouring the former set of ecosystem properties at the expense of the latter.

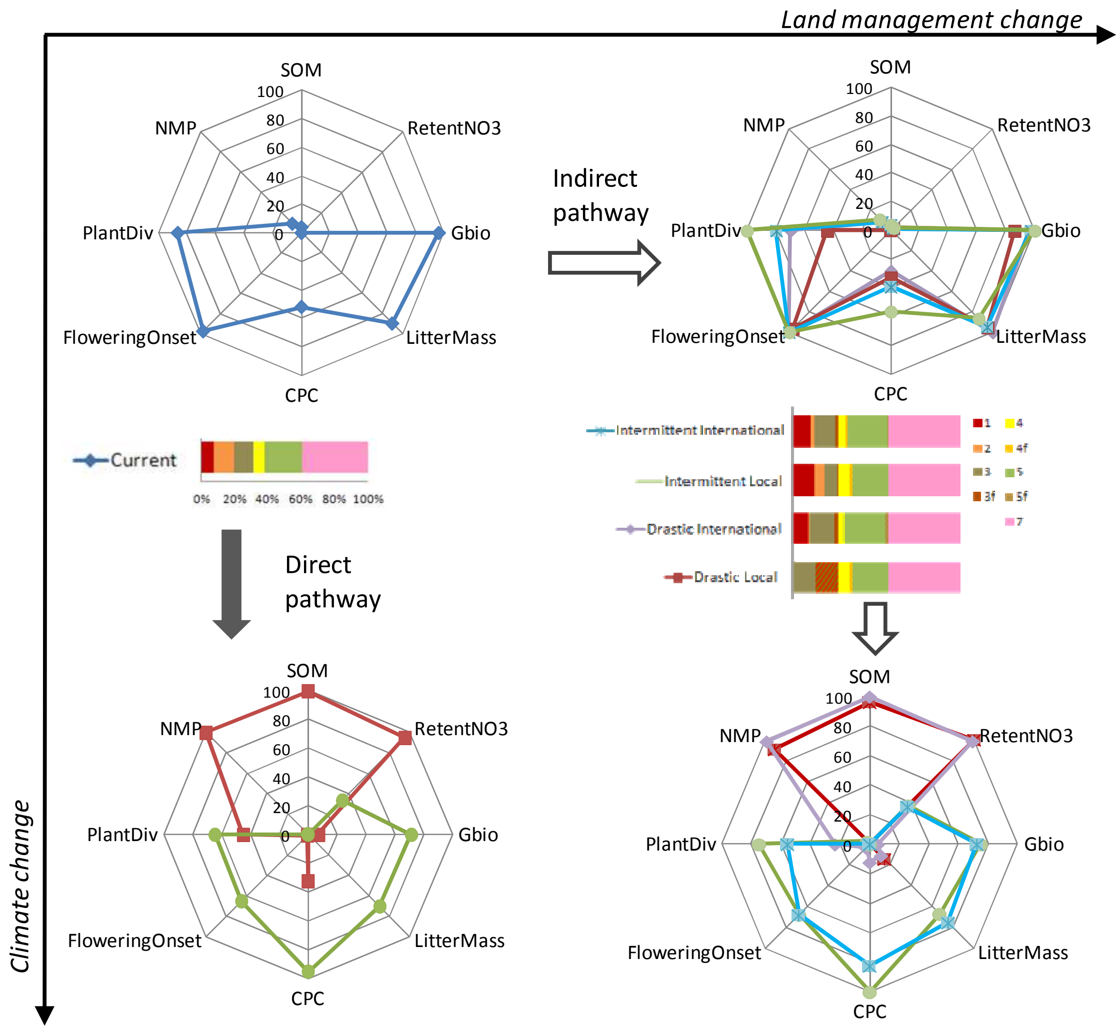


Figure 2: Illustration of ecosystem properties under different land management and climate scenarios. Standardized units along each axis indicate the condition of each ecosystem property. The top left diagram represents ecosystem properties in the current context. The bottom left diagram represents the direct pathway of climate effects considering the effect of climate only under the current land management configuration. The right side of the figure represents the indirect pathway of climate effects through adaptive land management with the top diagram representing the effects of land management change under different scenarios with current climate conditions and the bottom diagram representing the combination of both direct and indirect effects of climate. Stack bar graphs present the percentage of each land management type under the current situation and the four land management scenarios. Crude protein content, CPC; Green biomass, Gbio; Litter mass, LitterMass; Nitrate retention, Retent NO3; Soil organic matter content, SOM; Nitrogen mineralization, NMP; PlantDiv; Date of grass flowering onset, FloweringOnset.

3.2 Direct vs. indirect effects of climate on ecosystem properties and their bundles

Climate alternatives strongly influenced variations in most ecosystem properties (SOM, LeachNO3, Gbio, CPC,Litter, NMP and date of flowering onset) with more than 89% of variance explained by climate in ANOVAs (Table 2, though this effect was more moderate for plant diversity. Posthoc tests (results not shown) confirmed the previous patterns (Figure 2), showing that the drastic climate alternative significantly modified all ecosystem properties. NO3 retention, Litter and CPC were strongly changed in all land management scenarios in comparison to current land management, while SOM, Gbio and SoilFert change were significantly modified by the "Drastic-Local" land management scenario (Posthoc tests results not shown).

	Climate % variance			**Land management** % variance		
	L	**T**	**NT**	**L**	**T**	**NT**
Soil Organic Matter	0.99***	0.93***	0.99***	0***	0.06***	0.0***
NO3 retention	0.99***	0.66***	0.95***	0***	0.33***	0.04***
Green biomass production	0.99***	0.86***	0.98***	0***	0.13***	0.01**
Fodder crude protein content	0.89***	0.17**	0.70***	0.1*	**0.81*** **	0.29***
Litter mass	0.98***	0.68**	0.96***	0.02*	0.28*	0.04**
Date of grass flowering onset	0.99***	0.99***	0.99***	0.0***	0.01**	0.01**
Plant diversity	0.66***	0.20***	0.87***	0.33***	**0.80*** **	0.12***
Nitrogen mineralization potential	0.99***	0.58***	0.99***	0***	0.42***	0.0***

Table 2: Variance partitioning by ANOVA showing the % variance accounted by direct (Climate) and indirect (Land management) effects for different ecosystem properties. Results are presented for the entire landscape (L), or for terraced grasslands (T) and unterraced grasslands (NT) analysed individually. *P < 0.05; **P < 0.01; ***P < 0.001.

Redundancy analysis elucidated how co-variation in the set of eight ecosystem properties for the entire landscape varied according to drought and land management scenarios. The primary axis of differentiation among scenarios explained 84% of the total variance and represented direct climate effects (Figure 3). Simultaneous increases in nitrogen mineralization, soil organic matter and nitrate retention were strongly and positively related to drastic droughts, at the expense of the other properties, which were favoured under current climate or intermittent droughts (Figure 3). The second axis explained 10% of the total variance and represented contrasts across land management scenarios. When explicitly considering area under key management types or mowing and fertilization, as explanatory variables, this second axis contrasted scenarios with a greater area under fertilization (corresponding to the 'drastic –local' and 'drastic-international' land management scenarios – see also Figure 2, stacked bar graphs, land management types 1, 3f, 4f and 5f) from those favouring grazing against mowing (corresponding to the 'intermittent-local' land management scenario – see also Figure 2, stacked bar graphs, land management types 3, 3f, 5 and 5f) (Figure 3). Fodder crude protein content was separated from the other ecosystem properties on this second axis reflecting its positive responses to mowing.

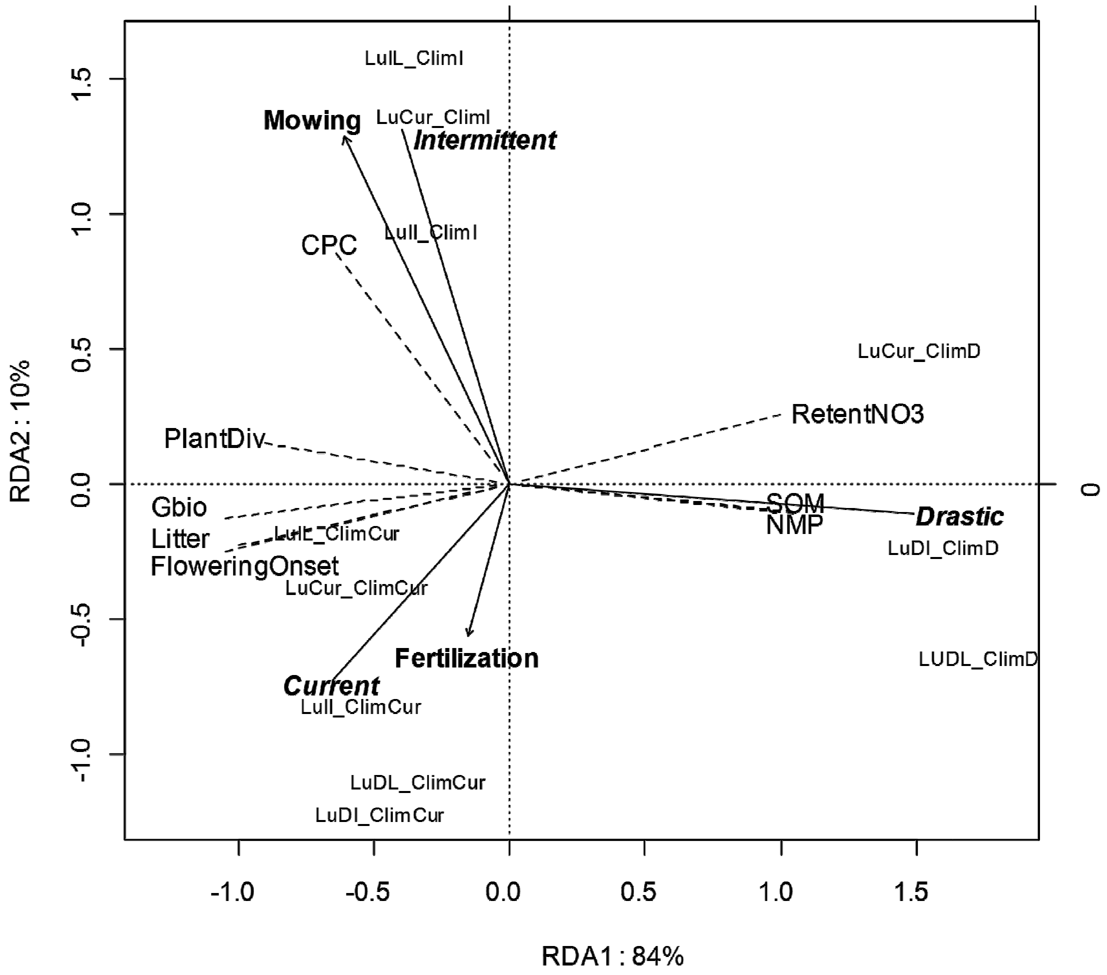


Figure 3: Responses of ecosystem properties for all combinations of scenarios combining climate and land use alternatives. Land use alternatives were characterized by % area under mowing and % area under fertilization. Scenario combinations and their acronyms are those presented in Table 1. Ecosystem properties are displayed in plain font and explanatory variables in bold. Crude protein content, CPC; Green biomass, Gbio; Litter mass, LitterMass; Nitrate retention, Retent NO3; Soil organic matter content, SOM; Nitrogen mineralization, NMP; Plant Diversity, PlantDiv; Date of flowering onset, FloweringOnset. All canonical axes were significant (Monte Carlo permutation test, *p* < 0.001).

3.3 Mechanisms underpinning direct and indirect effects of climate on ecosystem properties

The ANOVA for unterraced grasslands revealed the same variation of individual ecosystem properties as at landscape scale. In contrast, on terraces most of the variance in fodder crude protein content (81%) and in plant diversity (80%) was due to land management, while variation of the other ecosystem properties was still mainly driven by climate (> 50%). Individual RDAs for non-terraces and terraces presented the same results as for the entire landscape, with the first axis explaining variation due to climate (77% variance for non-terraces and 50% variance for terraces), and the second axis explaining variation due to land management (7% variance for non terraces and 29% variance for terraces).

Spider graphs of ecosystem properties by land management types revealed three patterns of ecosystem properties (Figure 4) related respectively to the three climate alternatives. These patterns were not affected by land management alternatives other by the addition of new management types associated with fertilization (3f, 4f, 5f) since these alternatives only affected the representation of different types across the landscape, and thereby the aggregate values of ecosystem properties at landscape scale (Figure 2). Therefore they represented the effects of changes in ecological parameters in response to climate.

Intermittent drought closely resembled current climate, with only a slight decrease in most ecosystem properties. In contrast, drastic drought resulted in a dramatic increase in NMP and SOM, and to a lesser increase in NO3 retention especially in terraces. There was a concomitant drastic reduction in production and litter, with earlier grass flowering onset, and some of in plant diversity especially in terraces (LMT 1, 2, 3) and in unmown unterraced grasslands (LMT 5).

Fertilization (LMT 1 vs. 2, 3f vs. 3, 4f vs. 4, 5f vs. 5) had heterogeneous effects on ecosystem properties across land management types and scenarios, although these tended to be stronger in unterraced grasslands. Overall, fertilization effects were often weaker and even opposite under intermittent drought as compared to current climate or drastic drought.

Although fertilization increased production in terraces (LMT1&3) and mown unterraced grasslands (LMT4), it resulted in a large loss in production in mown unterraced grasslands (LMT5) except under intermittent drought. The response of standing litter was similar to that of biomass production in unterraced grasslands (increases and decreases for LMT4 and LMT5 respectively), but it was opposite to that of biomass in terraces except under drastic drought. Increased production in terraces was accompanied by decreased CPC except under drastic drought, whereas in LMT4 it was associated with decreased CPC, except under intermittent drought. Conversely, decreased production in LMT5 was associated with increased CPC, except under intermittent drought. Fertilization effects on date of grass flowering onset were overall weak, and inconsistent across land management types and scenarios.

Fertilization had limited impacts on plant diversity in LMT3 except for an increase under intermittent drought, while it strongly decreased plant diversity in LMT4 except under intermittent drought and increased it in LMT1 and LMT5, with especially strong effects under drastic drought.

While, logically, nitrate retention decreased with fertilization across different management types - except mown terraces (LMT1), soil organic matter and nitrogen mineralization were mostly unchanged in terraces (T1&3) except under drastic drought where they decreased, but increased strongly in T5 and decreased in T4, except under intermittent drought for both cases.

Grazing as compared to mowing (LMT 1 and 2 vs. 3, 4 vs. 5) on the other hand had consistent effects across climate alternatives but varied across land management types. In terraced grasslands, grazing decreased biomass production, CPC, nitrogen mineralization, and strongly reduced plant diversity, while it increased standing litter and nitrate retention. In unterraced grasslands grazing increased biomass production and standing litter, while it decreased CPC, nitrogen mineralization, nitrate retention and strongly reduced plant diversity.

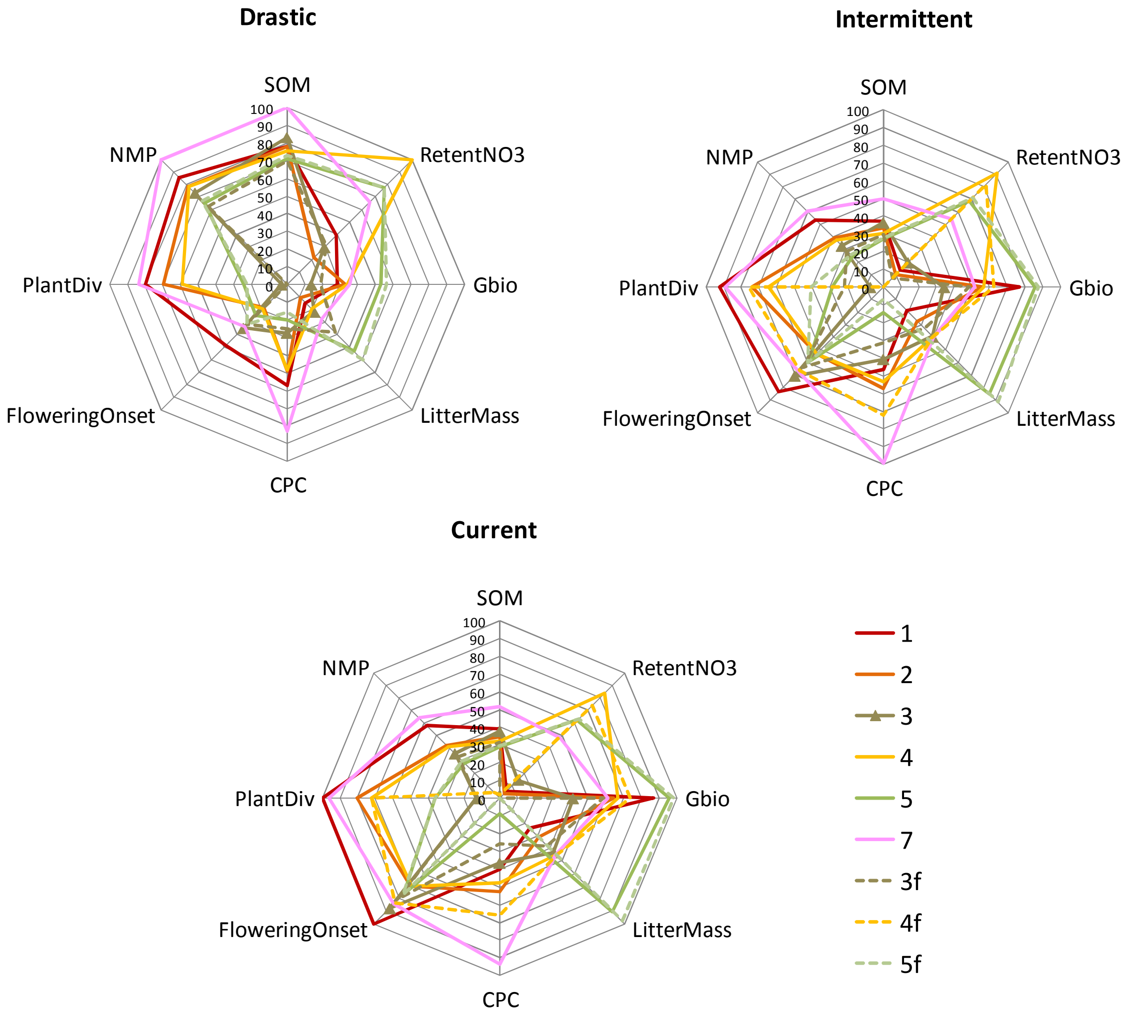


Figure 4: Bundles of ecosystem properties for the three climate alternatives. Each of the spider graphs represents the configuration obtained using the land management alternative with the greatest range of land management types (Intermittent-local land management scenario).

4 Discussion

While climate and land-use are the two main drivers of change in ecosystem services (Carpenter et al., 2006), to our knowledge this is the first trait-based study identifying direct and indirect effects of climate on ecosystem properties, and attempting to tease out underlying mechanisms associated with plant functional ecology and landscape patterns.

4.1 Direct and indirect effects of climate change on ecosystem properties underpinning ecosystem services

Our approach of apportioning variance in ecosystem properties to climate and land management effects showed that, at landscape scale, most of the ecosystem properties considered responded predominantly to climate. These included soil organic matter, nitrogen mineralization, nitrate leaching, biomass production, litter mass and date of grass flowering onset. In contrast a few ecosystem properties such as fodder crude protein content and plant diversity showed an intermediate situation with a joint influence by climate and management. The climate alternative with the greatest influence in ecosystem function variation was the drastic alternative representing periods of four consecutive years of drought. Whatever the land management, nitrogen mineralization, nitrate retention and soil organic matter increased significantly under drastic drought conditions, while all other properties severely decreased. Conversely, under the intermittent climate alternative, nitrogen mineralization, nitrate retention, soil organic matter and the other ecosystem properties all decreased slightly as compared to current climate. In the two last climate contexts, ecosystem properties except CPC, nitrogen mineralization and soil organic matter increased slightly in case of manuring.

Overall, it is however difficult to assess our results against previous studies, given the paucity of scenario-based ecosystem service assessments, and in particular the dearth of assessments using climate scenarios (Seppelt et al., 2011). Few published studies quantifying scenarios of ecosystem services under climate change have focused on the effects of combined climate and land use scenarios, but have not attempted to tease out their respective effects. At continental scale, Schröter et al. (2005) analysed how such scenarios impacted a broad array of ecosystem properties and services across Europe, but they did not attempt to tease out the relative effects of land use as compared to climate. Nevertheless for these scenarios, at European scale land use effects on patterns of plant diversity were negligible after climate change effects had been accounted for because at such coarse scale land use was primarily driven by climate (Thuiller et al., 2004).

At landscape scale, most scenario studies have focused on land cover change effects. For example, in the Swiss Alps the comparison across a temperature increase scenario, which translated as an increase in forest cover, and a land use scenario associated with tourism development and resulting urbanization demonstrated how impacts on avalanche protection, scenic beauty and wildlife habitat resulted from the scenarios' effects on these critical land cover elements (Grêt-Regamey et al., 2008). Likewise, in the Prairie Potholes Region of Dakota, impacts of alternative land use change scenarios on carbon sequestration, soil retention and waterfowl status were related to the area of land under conservation (Gascoigne et al., 2011).

These studies also considered impacts on one or several ecosystem services, but not on bundles and trade-offs among ecosystem services. This issue has been addressed at global and regional scales considering land use scenarios, but as far as we know, not explicitly combined scenarios of climate and land use. Land use scenario studies have compared bundles of ecosystem services, and associated synergies and antagonisms, across scenarios (Bohensky et al., 2006), or how such bundles may trade-off with biodiversity conservation across scenarios (Nelson et al., 2009). In our study, climate explained most of the variation in ecosystem properties, with substantial effects on the bundles of ecosystem properties across combined climate and land use change scenarios. However, synergies and antagonisms between ecosystem properties did not change across scenarios.

Finally, we considered the effects of specific management practices (e.g. fertilization and mowing) affecting the relationships among ecosystem properties rather than broad land use types as considered in most other spatially explicit studies of ecosystem services (Bennett et al., 2009). Even when studies have developed land use scenarios using advanced participative methods, ecosystem service impacts have been quantified using broad land cover categories (forest, cropland, grasslands, urban…) (Bayfield et al., 2008; Swetnam et al., 2011). Our analysis clearly demonstrated the value of considering explicit details of management practices as these were important explanatory variables teasing out land use alternatives.

4.2 Mechanisms underpinning climate change on ecosystem properties

Landscape scale effects of scenarios are an area weighted average of effects on each of the land management types, thus combining (1) changes in area covered by each land management type, i.e. in landscape composition, and (2) ecological effects of scenarios on ecosystem properties within each land management type. Thus the relative contributions of direct and indirect climate effects were related to two main causes discussed in the following sections: (1) the magnitude of changes in management across the landscape depending on area of different land use types, and (2) the relative magnitude of ecological effects of climate and management.

4.2.1 Effects of landscape composition

The surprising small effect of land management can be explained by the large area in the landscape occupied by unterraced grasslands and alpine meadows (more than 70% together, with 29% and 42% respectively) which incurred respectively few or no changes in the different land management scenarios (see stacked bar graphs in Figure 2). Therefore, the effects of land management change occurring in terraces were hidden by their small area as compared to the other grassland types. Nevertheless, while the redundancy analysis on terraces revealed a relatively greater role of land management in ecosystem properties variations (29% variance explained by land management change on terraces against 7% on unterraced grasslands), climate remained the main driver of change except for plant diversity and crude protein content. This is due to the fact that although terraces were concerned by the major changes (Figure 4), these changes were limited in most scenarios in terms of both land management types and relative area. The greatest impacts on ecosystem properties resulted from the conversion of mowing to grazing under the drastic-local scenario, which caused an increase of grazing on terraces from 12 to 17% of the total area (Lamarque et al) and an increase in fertilized areas of the same magnitude across the entire landscape.

Such small changes in land management even under the most severe scenario were due to constraints induced by the high mountain environment (topography, climate, short growing season, low productivity) (Mottet et al., 2006; von Glasenapp and Thornton, 2011) providing little possibility of diversification of practices and farming systems. More contrasted results may be expected in other mountain farming systems where less productive grasslands are abandoned and where conversion from more intensive artificial and fertilized grassland or crops (e.g. maize) to grasslands has occurred (Briner et al., 2012; Martin et al., 2011; Nettier et al., 2010) leading to a stronger direct drought effects on land management.

4.2.2 Functional mechanisms underpinning scenario effects on ecosystem properties and their bundles

Nitrogen is one of the most limiting elements for plant growth in subalpine grasslands (Robson et al., 2010). Available inorganic nitrogen is limited due to the cold, and some instances dry, climate and seasonal variability strongly constraining soil organic matter mineralization (Harrison et al, 2007). Nitrogen fertility determines the quality of plant live and dead material, and resource availability to soil microorganisms, thereby feeding back to carbon and nitrogen cycling (Zeller et al., 2000), Quétier et al., 2007, Robson et al., 2010, Grigulis et al., in review). In our models, ecosystem properties were driven by two core sets of variables relating to fertility (NNI and $NO_{3,}$ Table S2) and plant traits respectively. These variables were modified in scenarios either directly, such as the increased fertility by fertilization, the reduction in fertility by drastic drought, or the modifications of trait values in response to drastic drought (Klumpp and Soussana, 2009). Additionally trait values were modified indirectly following changes in fertility. Importantly, the two climate alternatives differed in the involvement of these two mechanisms, where under intermittent drought trait values were only modified as a result of changed fertility, and thus of changed management, whereas under drastic drought trait values both pathways were combined.

Drought effects on ecosystem properties associated mostly with microbial processes (soil organic matter, nitrogen mineralization potential and nitrate retention; Grigulis et al., in review) resulted from the reduction in microbial activities as reduced water availability slows down litter decomposition (Benot et al., in revision; Couteaux et al., 1995), and/or microbial nitrification and denitrification activities. These effects translated to increased carbon and nitrogen sequestration (Bardgett et al, 2005, Robson et al. 2010), and thus reduced availability to plants. This feedback was reflected in the changes in plant traits towards more conservative (greater LDMC, lower LNC) plant strategies and decreased plant height. Effects of these plant functional changes then cascaded to ecosystem properties driven by plant traits: biomass production, standing litter, as well as to crude protein content (Lavorel & Grigulis, 2012, Grigulis et al., in review).

Land management changes impacted ecosystem properties through their direct effects on nutrient availability and their indirect effects on plant traits (Quétier et al., 2007, Lavorel et al., 2011). Increased nutrient availability through manuring shifted communities from dominant conservative species (*e.g. Festuca paniculata*, *Bromus erectus*, *Sesleria caerulea*) to a more diverse array of species with an exploitative nutrient economy (e.g. *Dactylis glomerata, Agrostis capillaris, Trisetum flavescens*, *Heracleum sphondylum)* (Quetier et al., 2007). The main consequence of this functional shift was greater biomass production and reduced litter accumulation in terraced mown grasslands and unterraced mown grassland, but the opposite effects in unterraced unmown grasslands (Lavorel & Grigulis 2012). Conversion from mowing to extensive grazing indeed promoted dominance by species with conservative leaf traits (e.g. high LDMC), and in the case of unterraced grasslands, taller plants, especially *Festuca paniculata* which is promoted by grazing avoidance (Quétier et al., 2007).

The statistical models which we used to project ecosystem properties under the scenarios reflected their actual correlation structure of ecosystem properties and thus had direct consequences for expected bundles (Figure 5). Overall, observed relationships were consistent with expectations. The positive correlations between nitrate retention and biomass production or standing litter were opposite to expectations from their controlling traits, but could be explained by fertility effects on these variables, as well as flow of nutrients (litter accumulation promoting nutrient retention and biomass production uptaking nitrates from the soil and thereby reducing leaching).

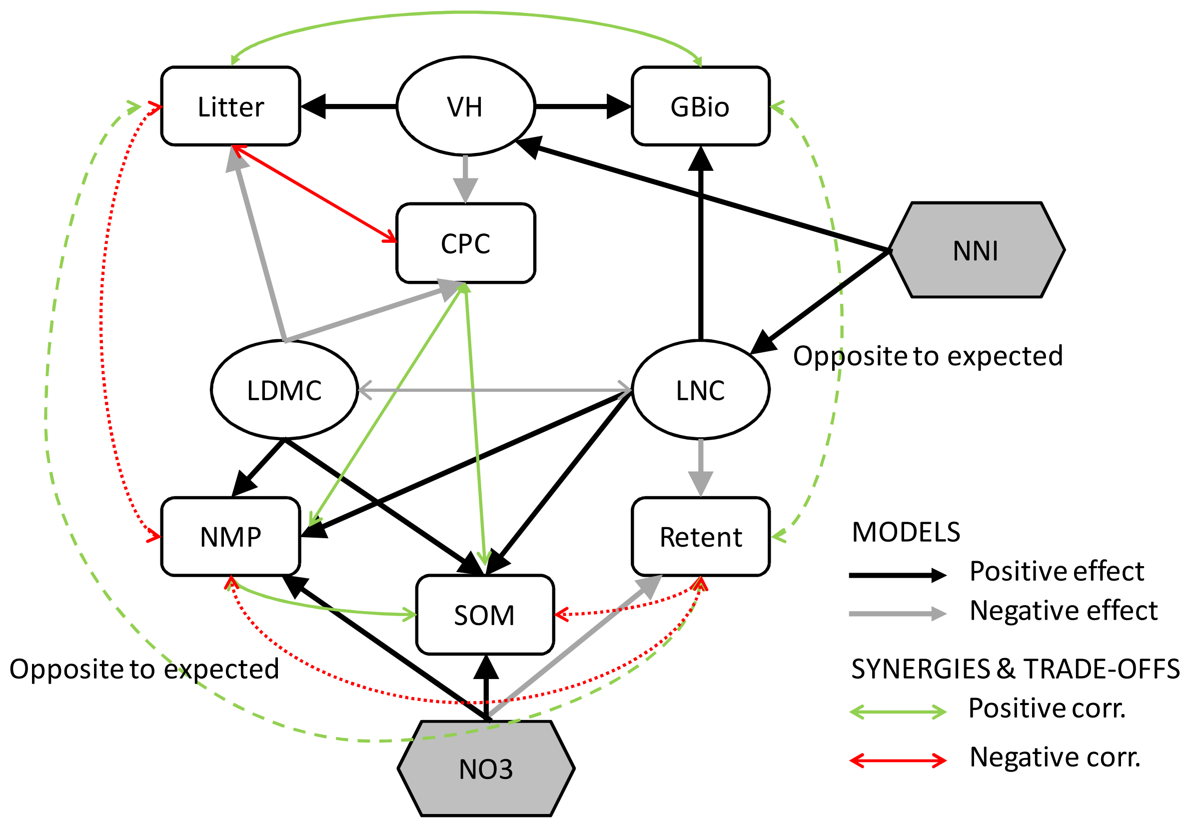


Figure 5: Traits based models of ecosystem properties and resulting correlations. Full double arrows show those correlations expected from the models that were actually verified at landscape scale. Dashed double arrows are correlations that were either opposite to expectations from the structure of models. The dotted double arrows represent an expected negative correlation between standing litter and NMP, and nitrate retention and NMP which were not verified.

Finally, the antagonistic relationship between standing litter and plant diversity was explained by the inhibitory effects of litter, including via light interception (Gross et al., 2010b) and rainfall interception resulting in enhanced drought effects (Gross et al., 2008). Synergies between plant diversity and both SOM and NMP were likely indirect effects of fertility, which both promoted species diversity (Quétier et al., 2007) and contributed to greater carbon and nutrient sequestration (Figure 2; Grigulis et al., in review).

Thus, consistent with the framework proposed by Bennett et al. (2009), the mechanisms underlying bundles of ecosystem properties and trade-offs in response to scenarios resulted both from common responses of different ecosystem properties to fertility parameters (NNI and NO3), and to interactions between ecosystem properties as a result of common driving traits.

4.3 Implications for ecosystem services management

Ecosystem services can be assessed by combining relevant ecosystem properties elicited as important by stakeholders (de Chazal et al., 2008; Lamarque et al., 2011a)(+ autres ref). Based on perception by stakeholders from the study site (Lamarque et al., 2011b; Quétier et al., 2010b) and the classification of ecosystem services by Zhang et al. (2007): (1) the regulation services of maintenance of soil fertility (NMP) and of soil organic matter (SOM)) were input services, (2) the marketed output could be quantified by the combination of biomass production (Gbio), fodder quality (CPC) and the date of grass flowering onset (FloweringOnset), and (3) non marketed output resulted from the combination of cultural value (conservation of plant diversity (PlantDiv), aesthetics (litter quantity)), and the regulation services of water quality (RetentNO3) and carbon storage (SOM)).

Overall, climate change, and especially drastic drought resulted in a very marked shift from production services to regulation services, and a reduction in cultural value due to the loss of plant species diversity. From an agronomic perspective (Zhang et al. 2007), input services supporting production were enhanced with the increase in the service of maintenance of soil fertility and in overall soil quality through increased soil organic matter content. While marketed production was directly reduced, a range of outputs that are currently not marketed were enhanced, such as carbon sequestration through increased SOM and increased water quality through nitrate retention. Maintaining farmer livelihoods under such a scenario would therefore require the establishment of systems of payment for these ecosystem services. Such payments could be attained under the ‘global’ land management alternative, but would be much less likely under the ‘local’ alternative as these services benefit to people outside of the production area (Costanza, 2008).

Given their limited impacts on ecosystem properties, land management changes per se only marginally affected the provision of ecosystem services. Their main impacts regarded the cultural value through decreased plant diversity resulting from the conversion from mowing to grazing, and the quality of production through fertilization effects on crude protein content. As opposed to crude protein content, decreased plant diversity has no direct impact on farmers’ income. Thus, conserving biodiversity requires financial incentives which encourage farmers to maintain mowing by compensating for high costs of this practice in a high mountain area. But to cope with unexpected events such as droughts, flexibility in grassland management is important for low input, extensively managed grasslands. In this context, payments based on results instead of actions, as most current European agri-environment schemes, are suggested as they would allow more flexibility in management (Schwarz et al., 2008). Therefore, policies should consider trade-offs among desired ecosystem services and give farmers means to reach the ecosystem services results required, considering climate change and its direct and indirect effects on ecosystem service provision.

Acknowledgments

We thank Aloïs Artaux, Pierre Gos, Coline Byczek and Emilie Crouzat for their useful advices. We acknowledge the Joseph Fourier Alpine Station for logistical support. This research was conducted on the long term research site "Zone Atelier Alpes", a member of the ILTER-Europe network and LTSER networks (ZAA publication n°XXXXX). This work was supported by ANR through the FP6 BiodivERsA Eranet VITAL project (ANR-08-BDVA-008), by the French Environment Ministry (MEEDDM) through the GICC-2 SECALP project, and by EU through VOLANTE, EU project FP7-ENV-2010-265104.

References

Agrimonde®, 2009, Agricultures et alimentations du monde en 2050 : scénarios et défis pour un développement durable. Note de synthèse. INRA et Cirad (ed.), Paris.

Alcamo, J., Kok, K., Busch, G., Priess, J. A., Eickhout, B., Rounsevell, M., Rothman, D. S., Heistermann, M., Lambin, E. F., Geist, H., 2006, Searching for the Future of Land: Scenarios from the Local to Global Scale

Land-Use and Land-Cover Change, Springer Berlin Heidelberg, pp. 137-155.

Anderson, B. J., Armsworth, P. R., Eigenbrod, F., Thomas, C. D., Gillings, S., Heinemeyer, A., Roy, D. B., Gaston, K. J., 2009, Spatial covariance between biodiversity and other ecosystem service priorities, *Journal of Applied Ecology* **46**(4)**:**888-896.

Ansquer, P., 2006, Caractérisation agroécologique des végétations prairiales naturelles en réponse aux pratiques agricoles. Apports pour la construction d'outils de diagnostic., Institut National polytechnique de Toulouse, Toulouse.

Barnaud, C., Trebuil, G., Dumrongrojwatthana, P., Marie, J., 2008, Area study prior to companion modelling to integrate multiple interests in upper watershed management of Northern Thailand, *Southeast Asian Studies* **45:**559-585.

Barnaud, C., Van Paassen, A. M., Trébuil, G., Promburom, T., Bousquet, F., 2010, Dealing with power games in a companion modelling process: lessons from community water management in Thailand highlands, *Journal of International agricultural and extension education* **16**(1).

Barreteau, O., Bots, P. W. G., Daniell, K. A., 2010, A Framework for Clarifying "Participation" in Participatory Research to Prevent its Rejection for the Wrong Reasons, *Ecology and society* **15**(2).

Barreteau, O., Bousquet, F., Attonaty, J. M., 2001, Role-playing games for opening the black box of multi-agent systems: method and lessons of its application to Senegal River Valley irrigated systems, *Jasss-the Journal of Artificial Societies and Social Simulation* **4**(2)**:**U75-U93.

Barreteau, O., Le Page, C., Perez, P., 2007, Contribution of simulation and gaming to natural resource management issues: An introduction, *Simulation & Gaming* **38**(2)**:**185-194.

Bayfield, N., Barancok, P., Furger, M., Sebastià, M. T., Domínguez, G., Lapka, M., Cudlinova, E., Vescovo, L., Ganielle, D., Cernusca, A., Tappeiner, U., Drosler, M., 2008, Stakeholder Perceptions of the Impacts of Rural Funding Scenarios on Mountain Landscapes across Europe. , *Ecosystems* **11:**1368-1382.

Bennett, E. M., Peterson, G. D., Gordon, L. J., 2009, Understanding relationships among multiple ecosystem services, *Ecology Letters* **12**(12)**:**1394-1404.

Benot, M. L., Saccone, P., Reydet, E., Vicente, R., Colace, M. P., Grigulis, K., Clément, J. C., Lavorel, S., in revision, Management, not summer climate manipulation, drives changes in biodiversity and functioning of a subalpine grassland, *Oecologia*.

Bigot, S., Rome, S., Biron, R., Laurent, J.-P., 2010, Mesures geophysiques (air-sol) a l'echelle d'une pelouse des hauts

plateaux du Vercors : analyse des variations hydroclimatiques locales et regionales (H. Q. *Colloque de l'Association Internationale de Climatologie* (ed O.P. Vincent Dubreuil, Valerie Bonnardot), ed.), Rennes, pp. pp. 77-82.

Boe, J., Terray, L., Habets, F., Martin, E., 2006, A simple statistical-dynamical downscaling scheme based on weather types and conditional resampling, *Journal of Geophysical Research-Atmospheres* **111**(D23).

Bohensky, E. L., Reyers, B., Van Jaarsveld, A. S., 2006, Future ecosystem services in a Southern African river basin: a scenario planning approach to uncertainty, *Conservation biology* **20**(4)**:**1051-1061.

Bohunovsky, L., Jager, J., Omann, I., 2010, Participatory scenario development for integrated sustainability assessment, *Regional Environmental Change* **11**(2)**:**271-284.

Borjeson, L., Hojer, M., Dreborg, K. H., Ekvall, T., Finnveden, G., 2006, Scenario types and techniques: Towards a user's guide, *Futures* **38**(7)**:**723-739.

Bossy, S., 1985, Associations foncières pastorales et groupements pastoraux : bilan d'une décennie *Revue de Géographie Alpine* **73**(4)**:**439-463.

Briner, S., Elkin, C., Huber, R., Gret-Regamey, A., 2012, Assessing the impacts of economic and climate changes on land-use in mountain regions: A spatial dynamic modeling approach, *Agriculture Ecosystems & Environment* **149:**50-63.

Bryant, C. R., Smit, B., Brklacich, M., Johnston, T. R., Smithers, J., Chiotti, Q., Singh, B., 2000, Adaptation in Canadian agriculture to climatic variability and change, *Climatic Change* **45**(1)**:**181-201.

Carpenter, S. R., Bennett, E. M., Peterson, G. D., 2006, Scenarios for ecosystem services: An overview, *Ecology and society* **11**(1)**:**14.

Carpenter, S. R., Mooney, H. A., Agard, J., Capistrano, D., DeFries, R. S., Diaz, S., Dietz, T., Duraiappah, A. K., Oteng-Yeboah, A., Pereira, H. M., Perrings, C., Reid, W. V., Sarukhan, J., Scholes, R. J., Whyte, A., 2009, Science for managing ecosystem services: Beyond the Millenium Ecosytem Assessment, *Proceedings of the National Academy of Science* **106**(5)**:**1305-1312.

Carpenter, S. R., Pingali, P. L., Bennett, E. M., Zurek, M. B., 2005, Ecosystems and Human Well-being: Scenarios. Volume 2: the Millenium Ecosystem Assessment. Washington DC.

.

Castella, J. C., Trung, T. N., Boissau, S., 2005, Participatory simulation of land-use changes in the northern mountains of Vietnam: the combined use of an agent-based model, a role-playing game, and a geographic information system, *Ecology and society* **10**(1).

Chan, K. M. A., Shaw, M. R., Cameron, D. R., Underwood, E. C., Daily, G. C., 2006, Conservation planning for ecosystem services, *PLoS Biology* **4**(11)**:**2138-2152.

Coreau, A., 2009, Dialogue entre des chiffres et des lettres. Imaginer et construire des futurs possibles en écologie, Université Montpellier II, pp. 524.

Costanza, R., 2008, Ecosystem services: Multiple classification systems are needed, *Biological Conservation* **141**(2)**:**350-352.

Couteaux, M. M., Bottner, P., Berg, B., 1995, Litter decomposition, climate and liter quality, *Trends in Ecology & Evolution* **10**(2)**:**63-66.

de Bello, F., Lavorel, S., Diaz, S., Harrington, R., Cornelissen, J. H. C., Bardgett, R. D., Berg, M. P., Cipriotti, P., Feld, C. K., Hering, D., da Silva, P. M., Potts, S. G., Sandin, L., Sousa, J. P., Storkey, J., Wardle, D. A., Harrison, P. A., 2010, Towards an assessment of multiple ecosystem processes and services via functional traits, *Biodiversity and Conservation* **19**(10)**:**2873-2893.

de Chazal, J., Quétier, F., Lavorel, S., Van Doorn, A., 2008, Including multiple differing stakeholder values into vulnerability assessments of socio-ecological systems, *Global Environmental Change* **18:**508-520.

de Jouvenel, H., 2002, La démarche prospective. Un bref guide méthodologique, *Futuribles* **247:**1-24.

Deboeuf, E., 2009, Adaptabilité des systèmes d'élevage de haute-montagne à des aléas. Le cas de Villar d'Arène, Enita de Clermont-Ferrand, France, pp. 91.

Diaz, S., Fargione, J., Stuart Chapin, F., Tilman, D., 2006, Biodiversity Loss Threatens Human Well-Being, *PLoS Biology* **4**(8)**:**1300-1305.

Diaz, S., Lavorel, S., de Bello, F., Quétier, F., Grigulis, K., Robson, T. M., 2007, Incorporating plant functional diversity effects in ecosystem service assessments, *Proceedings of the National Academy of Science* **104**(52)**:**20684-20689.

Duru, M., Felten, B., Theau, J., Martin, G., 2012a, A modelling and participatory approach for enhancing learning about adaptation of grassland-based livestock systems to climate change, *Regional Environmental Change***:**1-12.

Duru, M., Tallowin, J., Cruz, P., 2005, Functional diversity in low-input grassland farming systems: characterisation, effect and management, in: *Integrating Efficient Grassland Farming and Biodiversity* (R. Lillak, R. Viiralt, A. Linke, V. Geherman, eds.), pp. 199-210.

Duru, M., Theau, J. P., Cruz, P., 2012b, Functional diversity of species-rich managed grasslands in response to fertility, defoliation and temperature, *Basic and Applied Ecology* **13**(1)**:**20-31.

Egoh, B., Reyers, B., Rouget, M., Bode, M., Richardson, D. M., 2009, Spatial congruence between biodiversity and ecosystem services in South Africa, *Biological Conservation* **142:**553-562.

Engler, R., Randin, C. F., Thuiller, W., Dullinger, S., Zimmermann, N. E., AraÚJo, M. B., Pearman, P. B., Le Lay, G., Piedallu, C., Albert, C. H., Choler, P., Coldea, G., De Lamo, X., DirnbÖCk, T., GÉGout, J.-C., GÓMez-GarcÍA, D., Grytnes, J.-A., Heegaard, E., HØIstad, F., NoguÉS-Bravo, D., Normand, S., PuŞCaŞ, M., SebastiÀ, M.-T., Stanisci, A., Theurillat, J.-P., Trivedi, M. R., Vittoz, P., Guisan, A., 2011, 21st century climate change threatens mountain flora unequally across Europe, *Global Change Biology* **17:**2330-2341.

Etienne, M., 2003, SYLVOPAST: a multiple target role-playing game to assess negotiation processes in sylvopastoral management planning, *Jasss-the Journal of Artificial Societies and Social Simulation* **6**(2).

Foley, J. A., DeFries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., Chapin, F. S., Coe, M. T., Daily, G. C., Gibbs, H. K., Helkowski, J. H., Holloway, T., Howard, E. A., Kucharik, C. J., Monfreda, C., Patz, J. A., Prentice, I. C., Ramankutty, N., Snyder, P. K., 2005, Global Consequences of Land Use, *Science* **309**(5734)**:**570-574.

Garb, Y., Pulver, S., VanDeveer, S. D., 2008, Scenarios in society, society in scenarios: toward a social scientific analysis of storyline-driven environmental modeling, *Environmental Research Letters* **3**(4).

Garcia-Martinez, A., Bernues, A., Olaizola, A. M., Simulation of mountain cattle farming system changes under diverse agricultural policies and off-farm labour scenarios, *Livestock Science* **137**(1-3)**:**73-86.

Garnier, E., Cortez, J., Billes, G., Navas, M. L., Roumet, C., Debussche, M., Laurent, G., Blanchard, A., Aubry, D., Bellmann, A., Neill, C., Toussaint, J. P., 2004, Plant functional markers capture ecosystem properties during secondary succession, *Ecology* **85**(9)**:**2630-2637.

Gascoigne, W. R., Hoag, D., Koontz, L., Tangen, B. A., Shaffer, T. L., Gleason, R. A., 2011, Valuing ecosystem and economic services across land-use scenarios in the Prairie Pothole Region of the Dakotas, USA, *Ecological Economics* **70**(10)**:**1715-1725.

Gibon, A., 2005, Managing grassland for production, the environment and the landscape. Challenges at the farm and the landscape level, *Livestock Production Science* **96**(1)**:**11-31.

Gibon, A., Sheeren, D., Monteil, C., Ladet, S., Balent, G., 2010, Modelling and simulating change in reforesting mountain landscapes using a social-ecological framework, *Landscape Ecology* **25**(2)**:**267-285.

Girel, J., Quétier, F., Bignon, A., Aubert, S., 2010, Histoire de l'agriculture en Oisans. Hautes Romanche et pays faranchin. Villar d'Arène, Hautes-Alpes, in: *La Galerie de l'Alpe*, Station Alpine Joseph Fourier, Grenoble, France, pp. 79.

Grard, M., 2010, Le rôle des politiques publiques dans les services écosystémiques des prairies de montagne, Master sciences et politiques de l'environnement, UMPC-IEP, Paris.

Grassein, F., 2009, Mécanismes de variation des traits fonctionnels dans les prairies des Alpes, in: *Laboratoire d'ecologie alpine*, Université Jospeh Fourier, Grenoble, pp. 242.

Grêt-Regamey, A., Bebi, P., Bishop, I. D., Schmid, W. A., 2008, Linking GIS-based models to value ecosystem services in an Alpine region, *Journal of Environmental Management* **89**(3)**:**197-208.

Grigulis, K., Lavorel, S., Krainer, U., Legay, N., Baxendale, C., Dumont, M., Kastl, E., Arnoldi, C., Bardgett, R. D., Poly, F., Pommier, T., schloter, M., Tappeiner, U., Bahn, M., Clément, J. C., submitted, Combined influence of plant and microbial functional traits on ecosystem processes in mountain grasslands, *Journal of Ecology*.

Gross, N., Liancourt, P., Choler, P., Suding, K. N., Lavorel, S., 2010, Strain and vegetation effect on local limiting resources explain the outcome of biotic interactions, *Perspectives in Plant Ecology, Evolution and Systematics* **12:**9-19.

Gross, N., Robson, T. M., Lavorel, S., Albert, C., Le Bagousse-Pinguet, Y., Guillemin, R., 2008, Plant response traits mediate the effects of subalpine grasslands on soil moisture, *New Phytologist* **180:**652–662.

Haines-Young, R., Potschin, M., 2010, The links between biodiversity, ecosystem services and human well-being, in: *Ecosystem ecology: A New Synthesis* (D. Raffaelli, C. Frid, eds.), CUP, Cambridge.

Hooper, D. U., Chapin, F. S., Ewel, J. J., Hector, A., Inchausti, P., Lavorel, S., Lawton, J. H., Lodge, D. M., Loreau, M., Naeem, S., Schmid, B., Setala, H., Symstad, A. J., Vandermeer, J., Wardle, D. A., 2005, Effects of biodiversity on ecosystem functioning: A consensus of current knowledge, *Ecological Monographs* **75**(1)**:**3-35.

Jäger, J., Rothman, D., Anastasi, C., Kartha, S., van Notten, P., 2008, Training Module 6. Scenario development and analysis. GEO Resource Book. A training manual on integrated environmental assessment and reporting. Available at: http://www.unep.org/ieacp/iea/training/manual/module6.aspx.

Klumpp, K., Soussana, J. F., 2009, Using functional traits to predict grassland ecosystem change: a mathematical test of the response-and-effect trait approach, *Global Change Biology* **15**(12)**:**2921-2934.

Kremen, C., 2005, Managing ecosystem services: what do we need to know about their ecology?, *Ecology Letters* **8:**468-479.

Lamarque, P., Nettier, B., Barnaud, C., Artaux, A., Eveilleau, C., Dobremez, L., Lavorel, S., submitted, A participatory approach to map land management change based on the adaptive management of mountain livestock systems to drought and socio-economic scenarios, *Landscape and Urban Planning*.

Lamarque, P., Quetier, F., Lavorel, S., 2011a, The diversity of the ecosystem services concept and its implications for their assessment and management, *Comptes Rendus Biologies* **334**(5-6)**:**441-449.

Lamarque, P., Tappeiner, U., Turner, C., Steinbacher, M., Bardgett, R. D., Szukics, U., Schermer, M., Lavorel, S., 2011b, Stakeholder perceptions of grassland ecosystem services in relation to knowledge on soil fertility and biodiversity, *Regional Environmental Change* **11**(4)**:**791-804.

Lambin, E. F., Geist, H. J., Lepers, E., 2003, Dynamics of land-use and land-cover change in tropical regions, *Annual Review of Environment and Resources* **28:**205-241.

Lavorel, S., Garnier, E., 2002, Predicting changes in community composition and ecosystem functionning from plant traits: revisiting the Holy Grail, *Functional Ecology* **16:**545-556.

Lavorel, S., Grigulis, K., 2012, How fundamental plant functional trait relationships scale-up to trade-offs and synergies in ecosystem services, *Journal of Ecology* **100**(1)**:**128-140.

Lavorel, S., Grigulis, K., Lamarque, P., Colace, M.-P., Garden, D., Girel, J., Pellet, G., Douzet, R., 2011, Using plant functional traits to understand the landscape distribution of multiple ecosystem services, *Journal of Ecology* **99**(1)**:**135-147.

Lavorel, S. c., 2011, Adaptation des territoires alpins à la recrudescence des sécheresses dans un contexte de changement global (SECALP). Rapport de fin de contrat. LECA CNRS-Université J. Fourier Grenoble, Cemagref Grenoble, Parc National des Ecrins Gap.

Lemaire, G., Pflimlin, A., 2007, Les sécheresses passées et à venir: quels impacts et quelles adaptations pour les sysèmes fourragers, *Fourrages* **191:**163-180.

Lucas, E., 2010, Analyse, bilan et perspectives de gestion des pullulations de campagnols terrestres dans le secteur du Briançonais, Université Joseph Fourier, Grenoble 1, Grenoble.

MacDonald, D., Crabtree, J. R., Wiesinger, G., Dax, T., Stamou, N., Fleury, P., Gutierrez Lazpita, J., Gibon, A., 2000, Agricultural abandonment in mountain areas of Europe: Environmental consequences and policy response, *Journal of Environmental Management* **59:**47-69.

Martin, G., Felten, B., Duru, M., 2011, Forage rummy: A game to support the participatory design of adapted livestock systems, *Environmental Modelling & Software* **26**(12)**:**1442-1453.

MEA, 2005, Millennium Ecosystem Assessment. Ecosystems and Human Well-being: Synthesis, Island Press, Washington DC U.S.A.

Metzger, M. J., Rounsevell, M. D. A., Van den Heiligenberg, H., Perez-Soba, M., Hardiman, P. S., 2010, How Personal Judgment Influences Scenario Development: an Example for Future Rural Development in Europe, *Ecology and society* **15**(2).

Millennium Ecosystem Assessment, 2005, Ecosystems and human well-being: scenarios, Island Press, Washington D.C., USA.

Mora, O. c., 2008, Les nouvelles ruralités à l'horizon 2030. Des relations villes campagnes en émergence ?, Quae ed., Paris.

Moss, R. H., Edmonds, J. A., Hibbard, K. A., Manning, M. R., Rose, S. K., van Vuuren, D. P., Carter, T. R., Emori, S., Kainuma, M., Kram, T., Meehl, G. A., Mitchell, J. F. B., Nakicenovic, N., Riahi, K., Smith, S. J., Stouffer, R. J., Thomson, A. M., Weyant, J. P., Wilbanks, T. J., 2010, The next generation of scenarios for climate change research and assessment, *Nature* **463**(7282)**:**747-756.

Mottet, A., Ladet, S., Coque, N., Gibon, A., 2006, Agricultural land-use change and its drivers in mountain landscapes: A case study in the Pyrenees, *Agriculture Ecosystems & Environment* **114**(2-4)**:**296-310.

Naivinit, W., 2009, Modélisation d'accompagnement pour l'analyse des interactions entre usages des terres et de l'eau et migrations dans le bassin versant de la Lam Dome Yai au Nord Est de la Thaïlande, in: *Géographie humaine, économique et régionale*, Université Paris Ouest Nanterre-La Défense, pp. 344.

Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, D. R., Chan, K. M., Daily, G. C., Goldstein, J., Kareiva, P. M., Lonsdorf, E., Naidoo, R., Ricketts, T. H., Rebecca, M., 2009, Modeling multiple ecosystem services,biodiversity conservation, commodity production, and tradeoffs at landscape scales, *Frontiers in Ecology and Environment* **7**(1)**:**4-11.

Nettier, B., Dobremez, L., Coussy, J. L., Romagny, T., 2010, Attitudes of livestock farmers and sensitivity of livestock farming systems to drought conditions in the French Alps, *Revue De Geographie Alpine-Journal of Alpine Research* **98**(1)**:**383-400.

Niedrist, G., Tasser, E., Luth, C., Dalla Via, J., Tappeiner, U., 2009, Plant diversity declines with recent land use changes in European Alps, *Plant Ecology* **202**(2)**:**195-210.

Pachauri, R. K., 2007, Climate Change 2007: Synthesis Report. Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, IPCC.

Pagé, C., Terray, L., Boé, J., 2008, Projections climatiques à échelle fine sur la France pour le 21ème siècle : les scénarii SCRATCH08, in: *Technical Report TR/CMGC/08/64, Centre Européen de Recherche et de Formation Avancée en Calcul Scientifique (CERFACS)*.

Pak, M. V., Brieva, D. C., 2010, Designing and implementing a Role-Playing Game: A tool to explain factors, decision making and landscape transformation, *Environmental Modelling & Software* **25**(11)**:**1322-1333.

Pereira, H. M., Leadley, P. W., Proenca, V., Alkemade, R., Scharlemann, J. P. W., Fernandez-Manjarres, J. F., Araujo, M. B., Balvanera, P., Biggs, R., Cheung, W. W. L., Chini, L., Cooper, H. D., Gilman, E. L., Guenette, S., Hurtt, G. C., Huntington, H. P., Mace, G. M., Oberdorff, T., Revenga, C., Rodrigues, P., Scholes, R. J., Sumaila, U. R., Walpole, M., 2010, Scenarios for Global Biodiversity in the 21st Century, *Science* **330**(6010)**:**1496-1501.

Quéré, J. P., Garel, J. P., Rous, C., B., P., P., D., 1999, Estimer les dégâts du Campagnol terrestre en prairie naturelle, *Fourrages* **158:**133_147.

Quétier, F., Lavorel, S., Thuillier, W., Davies, I., 2007a, Plant-trait-based modelling assessment of ecosystem services sensitivity to land-use change, *Ecological Applications* **17**(8)**:**2377-2386.

Quétier, F., Rivoal, F., Marty, P., De Chazal, J., Lavorel, S., 2010a, Social representations of an alpine grassland landscape and socio-political discourses on rural development, *Regional Environmental Change* **10:**119-130.

Quétier, F., Rivoal, F., Marty, P., de Chazal, J., Thuiller, W., Lavorel, S., 2010b, Social representations of an alpine grassland landscape and socio-political discourses on rural development, *Regional Environmental Change* **10**(2)**:**119-130.

Quétier, F., Thebault, A., Lavorel, S., 2007b, Plant traits in a state and transition framework as markers of ecosystem response to land-use change, *Ecological Monographs* **77**(1)**:**33-52.

R Development Core Team, 2008, R: a language and envrionment for statistical computing (R Foundations for Statistical Computing, Vienna).

Raudsepp-Hearne, C., Peterson, G. D., Bennett, E. M., 2010, Ecosystem service bundles for analyzing tradeoffs in diverse landscapes, *Proceedings of the National Academy of Sciences* **107**(11)**:**5242-5247.

Reiss, J., Bridle, J. R., Montoya, J. M., Woodward, G., 2009, Emerging horizons in biodiversity and ecosystem functioning research, *Trends in Ecology & Evolution* **24**(9)**:**505-514.

Reyers, B., O'Farrell, P. J., Cowling, R. M., Egoh, B. N., Le Maitre, D. C., Vlok, J. H. J., 2009, Ecosystem Services, Land-Cover Change, and Stakeholders: Finding a Sustainable Foothold for a Semiarid Biodiversity Hotspot, *Ecology and society* **14**(1).

Rodriguez, J. P., Beard, T. D., Bennett, E. M., Cumming, G. S., Cork, S. J., Agard, J., Dobson, A. P., Peterson, G. D., 2006, Trade-offs across space, time, and ecosystem services, *Ecology and society* **11**(1).

Rouan, M., Kerbiriou, C., Levrel, H., Etienne, M., 2010, A co-modelling process of social and natural dynamics on the isle of Ouessant: Sheep, turf and bikes, *Environmental Modelling & Software* **25**(11)**:**1399-1412.

Rounsevell, M. D. A., Metzger, M. J., 2010, Developing qualitative scenario storylines for environmental change assessment, *Wiley Interdisciplinary Reviews-Climate Change* **1**(4)**:**606-619.

Rudmann-Maurer, K., Weyand, A., Fischer, M., Stocklin, J., 2008, The role of landuse and natural determinants for grassland vegetation composition in the Swiss Alps, *Basic and Applied Ecology* **9**(5)**:**494-503.

Sala, O. E., Chapin, F. S., Armesto, J. J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L. F., Jackson, R. B., Kinzig, A., Leemans, R., Lodge, D. M., Mooney, H. A., Oesterheld, M., Poff, N. L., Sykes, M. T., Walker, B. H., Walker, M., Wall, D. H., 2000, Biodiversity - Global biodiversity scenarios for the year 2100, *Science* **287**(5459)**:**1770-1774.

Schroter, D., Cramer, W., Leemans, R., Prentice, I. C., Araujo, M. B., Arnell, N. W., Bondeau, A., Bugmann, H., Carter, T. R., Gracia, C. A., de la Vega-Leinert, A. C., Erhard, M., Ewert, F., Glendining, M., House, J. I., Kankaanpaa, S., Klein, R. J. T., Lavorel, S., Lindner, M., Metzger, M. J., Meyer, J., Mitchell, T. D., Reginster, I., Rounsevell, M., Sabate, S., Sitch, S., Smith, B., Smith, J., Smith, P., Sykes, M. T., Thonicke, K., Thuiller, W., Tuck, G., Zaehle, S., Zierl, B., 2005, Ecosystem service supply and vulnerability to global change in Europe, *Science* **310**(5752)**:**1333-1337.

Schwarz, G., Moxey, A., McCracken, D., Huband, S., Cummins, R., 2008, An analysis of the potential effectiveness of a Payment-by-Results approach to the delivery of environmental public goods and services supplied by Agri-Environment Schemes, Macaulay Institute, Pareto Consulting and Scottish Agricultural College. , Report to the Land Use Policy Group, UK,, pp. 108.

Steffen, W., Crutzen, P. J., McNeill, J. R., 2007, The Anthropocene: Are Humans Now Overwhelming the Great Forces of Nature, *AMBIO: A Journal of the Human Environment* **36**(8)**:**614-621.

Strijker, D., 2005, Marginal lands in Europe - causes of decline, *Basic and Applied Ecology* **6**(2)**:**99-106.

Suding, K. N., Goldstein, L. J., 2008, Testing the Holy Grail framework: using functional traits to predict ecosystem change, *New Phytologist* **180**(3)**:**559-562.

Swetnam, R. D., Fisher, B., Mbilinyi, B. P., Munishi, P. K. T., Willcock, S., Ricketts, T., Mwakalila, S., Balmford, A., Burgess, N. D., Marshall, A. R., Lewis, S. L., 2011, Mapping socio-economic scenarios of land cover change: A GIS method to enable ecosystem service modelling, *Journal of Environmental Management* **92**(3)**:**563-574.

Thuiller, W., Araujo, M. B., Lavorel, S., 2004, Do we need land-use to model species distributions in Europe?, *Journal of Biogeography* **31:**353-361.

van Vliet, M., Kok, K., Veldkamp, T., 2010, Linking stakeholders and modellers in scenario studies: The use of Fuzzy Cognitive Maps as a communication and learning tool, *Futures* **42**(1)**:**1-14.

Vittoz, P., Randin, C., Dutoit, A., Bonnet, F., Hegg, O., 2009, Low impact of Climate change on subalpine grasslands in the Swiss Nothern Alps, *Global Change Biology* **15**(209-220).

Voinov, A., Bousquet, F., 2010, Modelling with stakeholders, *Environmental Modelling & Software* **25**(11)**:**1268-1281.

Volkery, A., Ribeiro, T., Henrichs, T., Hoogeveen, Y., 2008, Your Vision or My Model? Lessons from Participatory Land Use Scenario Development on a European Scale, *Systemic Practice and Action Research* **21**(6)**:**459-477.

von Glasenapp, M., Thornton, T., 2011, Traditional Ecological Knowledge of Swiss Alpine Farmers and their Resilience to Socioecological Change, *Human Ecology* **39**(6)**:**769-781.

Walz, A., Lardelli, C., Behrendt, H., Grêt-Regamey, A., Lundström, C., Kytzia, S., Bebi, P., 2007, Participatory scenario analysis for integrated regional modelling, *Landscape and Urban Planning* **81**(1-2)**:**114-131.

Washington-Ottombre, C., Pijanowski, B., Campbell, D., Olson, J., Maitima, J., Musili, A., Kibaki, T., Kaburu, H., Hayombe, P., Owango, E., Irigia, B., Gichere, S., Mwangi, A., 2010, Using a role-playing game to inform the development of land-use models for the study of a complex socio-ecological system, *Agricultural Systems* **103**(3)**:**117-126.

Zeller, V., Bahn, M., Aichner, M., Tappeiner, U., 2000, Impact of land-use change on nitrogen mineralization in subalpine grasslands in the Southern Alps, *Biological Fertility of Soils* **31:**441-448.

Zhang, W., Ricketts, T. H., Kremen, C., Carney, K., Swinton, S. M., 2007, Ecosystem services and dis-services to agriculture, *Ecological Economics* **64:**253-260.

Supplementary materials

Climate alternative	Drastic	Intermittent
	NNI or PNI	
	-50%	- 10%
	NNI or PNI response to manuring	
LMT1 -> LMT3f	(LMT3+LMT1)/2 + 10	(LMT3+LMT1)/2 + 20
LMT3 -> LMT3f	+ 10	+ 20
LMT4 or LMT5-> LMT4f	+ 5	+ 10
LMT4 or LMT5 -> LMT5f	+ 5	+ 10
	Species traits	
Vegetative height	x 0.5	+0.23*ΔNNI
LNC	2/3 in fertilised terraces x 0.8 in other LMT	+ 0.26*ΔNNI
LPC	2/3 in fertilised terraces x 0.8 in other LMT	+0.0042*ΔNNI
LDMC	x 1.2	unchanged

Table S1: Modifications in model parameters in drastic and intermittent climate scenarios. LMT, Land management types (see section 2 study site for more details); LNC, Leaf nitrogen concentration; LPC, Leaf phosphorus concentration; LDMC, Lead dry matter content; NNI, Nitrogen nutrition index; PNI, Phosphorous nutrition index.

	Cst	Alt	CWM VgH	FD VgH	CWM LDMC	CWM LNC	CWM LPC	FD LPC	NNI	pH	WHC	Soil NO3	V max	DEA	Simpson
Ecosystem property															
Gbio	-2		6.566			7.53					7.83				
Litter	372.3		3.19	-101.6	-0.47		-86.9	108.5							
CPC	201.9		-2.01		-0.27						4.6				
Plant diversity															X
Date of flowering onset	X														
LeachNO3	-0.78											0.38	0.77		
SOM (10^)	1.28				1.49 * log10 (LDMC/1000)									0.44	
NMP	1.37				1.92 * log10 (LDMC/1000)									1.02	
Soil parameters															
Soil NO3	0.10					0.04									
V max	0.001									0.26		0.57			
DEA	0.46								0.53			0.21			

Table S2: Summary of statistics from General linear models of ecosystem properties from abiotic variables and functional diversity components, traits community weighted mean (CWM) and functional divergence (FD). Cst, model intercept. Altitude, Alt (m). Plant vegetative traits: Leaf nitrogen concentration, LNC (mg g-1), Leaf phosphorus concentration, LPC (mg g-1); Vegetative height, VgH (cm). Ecosystem properties: Crude protein content, CPC (g/kg); Green biomass, Gbio (g/m²); Litter mass, Litter, (g/m²); Nitrate retention RetentNO3, (µg.g^{-1} soil); Soil organic matter content, SOM (%), Nitrogen mineralization, NMP (NH4-N µg.g^{-1}.d^{-1}) Soil parameters: Soil nitrate concentration, SoilNO3 (log) (µg.g^{-1} soil) ; Maximum nitrification rate, Vmax (µg N-NH4+.g-1.h-1), Potential denitrification enzyme activity, DEA (µg N-N2O.g^{-1}.h^{-1}).

	LuCur_ClimI	LuCur_climD	LuDI_ClimCur	LuII_cliCur	LuIL_climCur	LuDL_ClimCur	LuDI_climD	LuDL_climD	LIL_climI	LUII_climI	LuCur_cliCur
Litter-Gbio	0.7	0.8	0.7	0.7	0.7	0.9	0.9	0.8	0.7	0.7	0.7
CPC-SOM	0.9	0.9	0.9	0.8	0.8	0.8	0.9	0.8	0.8	0.8	0.9
CPC-Litter	-0.5		-0.5	-0.5	-0.5	-0.5			-0.5	-0.5	-0.5
PlantDiv-Litter	-0.6	-0.5	-0.5	-0.5	-0.5				-0.6	-0.5	-0.6
PlantDiv-SOM	0.7	0.6	0.7	0.6	0.6	0.7	0.7	0.7	0.6	0.6	0.6
PlantDiv-CPC	0.8	0.9	0.8	0.8	0.7	0.9	0.9	0.9	0.8	0.8	0.8
NMP- CPC	0.9	0.9	0.9	0.8	0.8	0.9	0.9	0.9	0.8	0.8	0.9
NMP-SOM	1	1	1	0.9	0.9	1	1	1	0.9	0.9	1
NMP-plantDiv	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8
Gbio-RetentNO3		0.7	0.5	0.5		0.7	0.9	0.8		0.5	
Litter-RetentNO3	0.7	0.6	0.7	0.7	0.7	0.6	0.7		0.7	0.7	0.7

Table S3: Significant correlations between pairs of ecosystem services having Pearson coefficient ≥ 0.5. Red signifies positive correlations, and blue signifies negative correlations.

Synthèse partie II

Cette deuxième partie avait pour but de répondre à la question : Comment la gestion des prairies affecte la fourniture de services écosystémiques dans un contexte de changement global ?

Cette deuxième partie a étudié l'influence de l'utilisation du sol sur la fourniture de services écosystémiques dans différents contextes climatiques et socio-économiques (chapitre 5 et 6). Une démarche participative impliquant 16 experts régionaux en agriculture et environnement a débouché sur la construction de 4 scénarios contrastés combinant climat (récurrence de différents types de sécheresse) et contexte socio-économique (Chapitre 5). Les scénarios ont été développés à l'échelle régionale et déclinés au niveau du site d'étude avec l'aide d'experts travaillant spécifiquement sur la zone du Lautaret. Ces scénarios décrits par des récits détaillés (*storylines*) donnent des variations d'éléments concrets comme par exemple la perte de rendement fourrager ou les changements d'abondance d'espèces importantes. Pour obtenir des cartes d'utilisation du sol pour chaque scénario, un jeu spatialisé a permis de mettre en situation les 8 éleveurs du site d'étude, d'étudier leurs adaptations face aux scénarios et de quantifier les changements de gestion qui en découlent. Ce jeu et les étapes de validation des données (entretiens individuels avec les éleveurs et des techniciens agricoles ; vérification numérique par méthode statistique) m'a permis de produire les cartes d'utilisation du sol à l'horizon 2030. Les principales adaptations identifiées sont l'arrêt partiel ou total de la fauche et la fertilisation de prés non fertilisés actuellement. Plus précisément, c'est dans le scénario climatique drastique que les surfaces fauchées sont le plus réduites, avec une réduction d'un facteur trois environ et des surfaces ne représentant plus que 8 % et 12 % du paysage, respectivement dans les scénarios socio-économiques locaux et internationaux. Les surfaces fertilisées sont quant à elles quasi doublées pour atteindre environ 15 % du paysage dans les quatre combinaisons de scénarios climatiques et socio-économiques.

D'un point de vue méthodologique, j'ai pu tester l'utilité et la validité de ma méthode participative de développement des scénarios incluant un jeu de rôles, ainsi que décrire l'évolution des trajectoires d'utilisation du sol selon le scénario et leurs conséquences pour l'hétérogénéité des paysages.

L'objectif final était d'en déduire des projections des services écosystémiques selon les scénarios (Chapitre 6). Au total 8 propriétés des écosystèmes (matière organique du sol, minéralisation potentielle de l'azote, lessivage des nitrates, date de floraison, quantité de biomasse aérienne, quantité de litière, matière azotée totale, diversité floristiques) ont été modélisées à l'aide des résultats d'études de terrains menées dans le cadre du projet VITAL et des modèles statistiques spatialement explicites développés pour cartographier les services actuels (Partie I : chapitre 4).

Ces modèles ont été ensuite appliqués aux cartes d'utilisation du sol associées à chaque scénario, avec des valeurs de traits fonctionnels calculées pour les changements d'abondance des espèces dominantes des différents types de prairies (trajectoires d'utilisation du sol) en fonction des sécheresses et des nouveaux modes de pratiques (fertilisation) à dire d'experts. Les propriétés des écosystèmes ont été traduites en services des écosystèmes (stockage de carbone, fertilité du sol, qualité des eaux, quantité de fourrage, qualité du fourrage, esthétique) selon les bénéfices retirés par les acteurs locaux (Partie I : chapitre 3). La démarche conceptuelle développée a permis de quantifier la contribution des effets directs et indirects du climat au travers de l'adaptation des éleveurs et des modifications de pratiques sur la fourniture de services. Les résultats indiquent que les services des écosystèmes à l'échelle du paysage varient principalement sous l'influence directe du climat car les modifications de pratiques n'affectent qu'une faible proportion du paysage. La modalité climatique « choc » dans laquelle quatre années de sécheresses de printemps consécutives ont lieu tous les 5 ans, en comparaison à la modalité « intermittente » où les sécheresses n'ont lieu qu'un an sur deux, tend à diminuer de manière conséquente la production de l'ensemble des services à l'échelle du paysage à l'exclusion du stockage de carbone, de la fertilité du sol et de la qualité des eaux (réduction des pertes en nitrates). Toutefois, l'ensemble des services réagissant de la même manière aux effets directs et indirects du climat, il n'y a pas de changement de relations entre les services. Ces relations constantes d'un scénario à l'autre regroupent à la fois des synergies et des compromis, au nombre de 7 et 5 respectivement. Les compromis les plus marqués sont la fertilité du sol versus la quantité de litière, et la diversité floristique versus la quantité de litière. Ces relations entre services s'expliquent par le fait que le climat et les pratiques affectent tous deux l'abondance des espèces exploitatrices ou conservatrices. Or, le comportement des traits entre ces groupes d'espèces se répercute sur les services que ces traits sous-tendent.

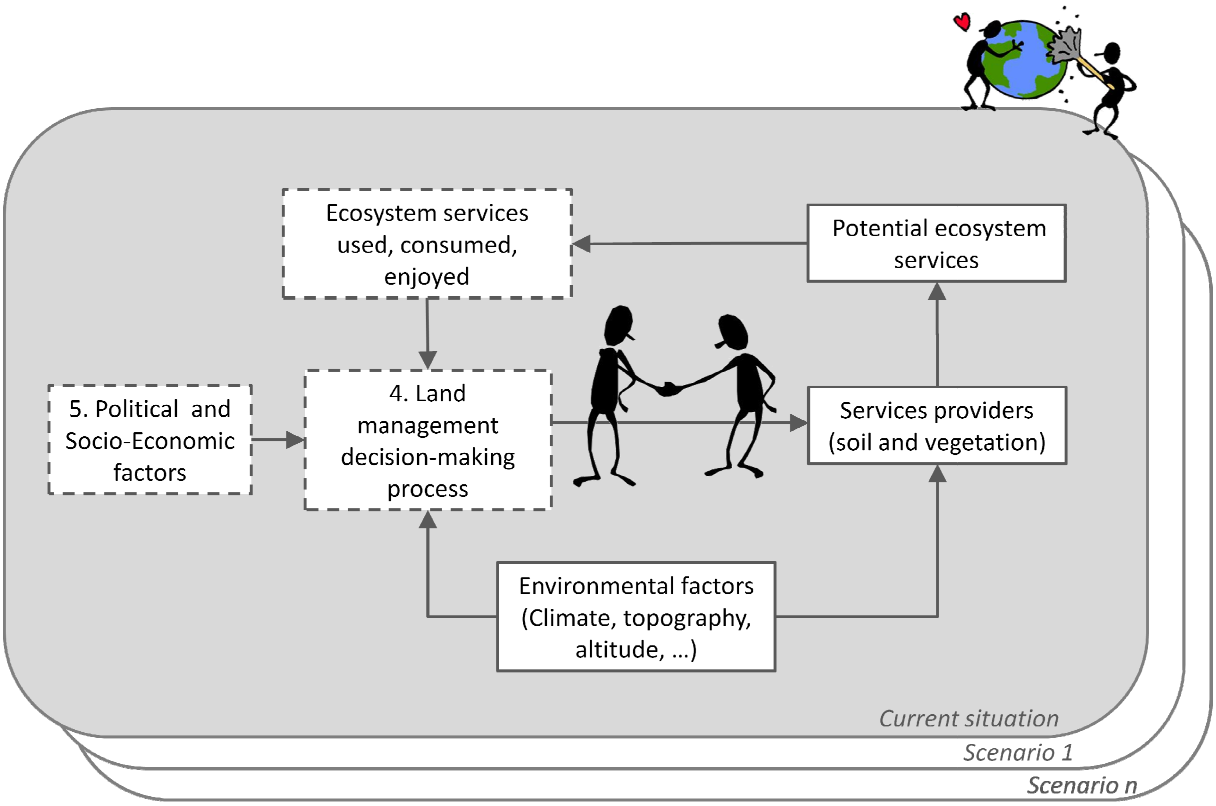
Partie III

De l’écosystème à l’homme

et

de l’homme à l’écosystème

Effets de rétroaction



Chapitre 7

Prise en compte des services écosystémiques dans les décisions des agriculteurs et effets de rétroaction des changements[13]

Abstract

The ecosystem services concept (ES) has emerged and spread widely recently, to enhance the importance of protecting ecosystems from global change in order to maintain their benefits for human well-being. Nevertheless, while the number of studies considering various dimensions of the interactions between ecosystems and land use via ES has been steadily increasing, integrated research addressing the complete feedback loop between biodiversity, ES and land use has remained mostly theoretical. Few studies consider the feedbacks from ecosystems to land use systems through ES, exploring how ES are taken into account in land management decisions. To fill this gap, we carried out a role-playing game (RPG) to explore how ES cognition mediates feedbacks from environmental change on farmers' behaviours in a mountain grassland system. On a close to real landscape game board, farmers were faced with changes in ES under climatic and socio-economic scenarios so as to prompt them to plan for the future and to take land management decisions where they deem necessary. RPG outcomes were complemented with additional agronomic and ecological data from interviews and fieldwork. This case-study demonstrates the effect of changes in ES on decision is mainly direct (direct feedback), without affecting knowledge and values. This occurs with changes farmers are used to face with, such as a reduction of forage quantity following droughts, which leads famers to grazing instead of mowing. Sometimes, ES cognitions are affected by ES changes or by external factors, the feedback is thus considered as indirect. This happened when fertilization was stopped after farmers learned that it was inefficient in a drought context. Our results also show that farmers' behaviour does not always reflect their attitudes towards ES because other factors including topographic constraints, social value of farming or farmer individual and household characteristics may be at stake when it comes to take land-management decisions.

[13] This chapter is a draft of a paper in preparation for Ecology and society : Lamarque P, Meyfroidt P, Nettier B, Barnaud C and Sandra Lavorel, Ecosystem services in French mountain farmers' decision-making.

1 Introduction

The Millennium Ecosystem Assessment (MEA, 2005) stressed the importance of assessing the consequences of ecosystem change on the services and goods that ecosystems provide to human well-being. Ecosystem contributions to human well-being in semi-natural ecosystem such as agro-ecosystems depend on the behaviour of the land manager, as changes in land uses and land management directly influence the ecosystem services delivered to multiple beneficiaries (Foley et al., 2005). Ecosystem services (ES), being defined as the outputs of ecosystems (called ecosystem functions (Lamarque et al., 2011a), chapter 2) from which people derive benefits (MEA, 2005), stress the need to integrate ecological and social science to study coupled human and natural systems (or social-ecological systems or human-environment systems, Collins et al., 2011; Diaz et al., 2011), and therefore require to explicitly address the complex feedback loops formed by reciprocal interactions between people and nature (Liu et al., 2007). “Feedbacks are not mechanistic, but depend on how environmental change affects ecosystem services and how human agents perceive the state of the system” (Meyfroidt, 2012). Nevertheless, while the number of studies considering various dimensions of the interactions between ecosystems and land use via ES has been steadily increasing, integrated research addressing the complete feedback loop between biodiversity, ES and land use has remained mostly theoretical. Indeed, most case-studies (e.g (Hein et al., 2006; Mace et al., 2011)) look at the interactions between ecosystems, ecosystem services, goods and human well-being by considering values generated for people, and close the loop by exploring changes and future trends in ecosystem services according to scenarios, with possible institutional responses. The full cascade of ecosystem services from processes to benefits (for a detailed explanation see, chapter 2, Lamarque et al., 2011a) is sometimes considered (e.g. (Mace et al., 2011)) but the feedbacks effects from ecosystem services values to human actions and their effects on ecosystem processes are rarely taken into account (Meyfroidt, 2012). Ecosystem services considerations in decision-making are part of two different areas of research in ecosystem services literature. Firstly, payments for ecosystem services consist in developing financial incentives to sustain appropriate management of resources so as to maintain or enhance ecosystem services delivery (Engel et al., 2008). Secondly, economic valuation (monetary or not) is used to make decision-makers aware of the importance of ecosystem services through the costs associated to their loss (Costanza et al., 1997; TEEB, 2009). However, the choice people make about how to use and to manage their environment is affected by the way people perceive ecosystems and their ability to provide values (Mace et al., 2011). In addition, psychology, decision sciences and behavioural economics show that individuals are not merely utility maximizer or financially rational (St John et al., 2011), and individual preferences are constantly evolving (Kumar and Kumar, 2008). These “complexities that lie in human attitudes, motivational systems and their behavioural manifestations are not adequately addressed by economic valuation methods and techniques”

(Kumar and Kumar, 2008). Recent reviews (Jones et al., 2011; Meyfroidt, 2012) point out the interest of using mental models to explore mechanisms through which individual decisions are made and enhance effective management of land and natural resources. Some studies have already looked at stakeholders' perception (Chapter 3)(Lamarque et al., 2011b; Lewan and Soderqvist, 2002; O'Farrell et al., 2007) or preferences and values (Duguma and Hager, 2011) in terms of ecosystem services. Other studies have addressed the question of farmers' decision-making process (Feola and Binder, 2010; Nettier et al., 2011), some of them taking into account interactions between cognitive components and actions (Lauer and Aswani, 2010; Tengö and Belfrage, 2004), but none of these focus on ES

To fill this gap, as well as the lack of empirical studies on feedback effects, this paper studies how ecosystem services are taken into account in land-use decisions in the context of mountain grasslands management. We chose a study area in the Central French Alps typical of extensive management systems found in drier European mountains, which is mainly composed of permanent grasslands used for livestock farming, i.e. a single type of land-use. Thus, we considered specifically on land-management decisions rather than on land-use decisions, and our analysis focused on behaviour of farmers since they are the key decisional actors in this system. Previous studies identified three main types of land management change in these systems that affect ecosystem services (Chapter 5)(Nettier et al., 2010; Quétier, 2006): (1) manuring *versus* not, (2) mowing *versus* grazing, (3) early *versus* late mowing. We tested the hypothesis that these three land management behaviours are driven by farmers' willingness to benefit from ecosystem services. Previous studies on farmers' behaviour have stressed the need to consider multiple potential explanatory factors (e.g. biophysical, economic, political) and the relationships among them in order to cope with the complexity of social-ecological system (Feola and Binder, 2010), which led us to consider the influence of multiple ecosystem services as well as a broader context of climate and socio-economic change. We did not aim at producing a new theoretical framework alongside with existing models of individuals or farmers' behaviour (see Meyfroidt, 2012, Feola and Binder, 2010; Jones et al., 2011), but rather to adapt the theoretical frameworks of Meyfroidt (2012) and Vignola et al. (2010) in order to explore the feedbacks between ecosystem services and behaviours through farmers' cognitions. In the following we first describe the cognitive model underpinning our analysis. We then present the methodology used to document how ecosystem services are taken into account in farmers' decisions and describe results for each component of the cognitive model. The discussion explores the feedback loop between ecosystem services and land-use through farmers' cognitive processes.

2 Conceptual framework

Figure 1 shows our conceptual model to analyse the cognitive process of ecosystem services feedback on farmers' behaviour. In this model, land-management behaviours result from decisions allowed or hindered by contextual factors. Decisions are themselves determined by cognitive factors regarding ecosystem services and contextual factors. Thus, (B = f(D, C)) and (D = f(K, V, C)), where:

-Behaviour (B) refers to an action or series of actions (here the land-use/ agricultural practices) selected among possible alternatives (Feola and Binder, 2010). Behaviours follow decisions (D) except when contextual factors (C) force the agent to make his/her actions deviate from the preferred alternative;

- Decisions (D) refer to the preferred action selected among alternatives, taking into account the knowledge (K) and values (V) about ecosystem services, as well as the influence of contextual factors (C).

- Knowledge (K) focuses specifically on farmers' knowledge about contributions of ecosystem components and dynamics to ecosystem services, and on effects of their practices on these ecosystem dynamics;

- Values (V) correspond to very general assessments about things (e.g. ecosystem services, income) that are seen as desirable (Dietz et al., 2005). Value can refer to economic value (use or non-use value) or moral value (Kumar and Kumar, 2008). In this study we refer to values attributed to ecosystem services;

- Contextual factors (C) are external factors (out of farmers' cognition) that can influence the decisions by affecting the valence attributed to the different options, or the behaviours by making an action easier or more difficult to carry out. They are for example, the climatic conditions, the social context of the agents or the political context (Feola and Binder, 2010).

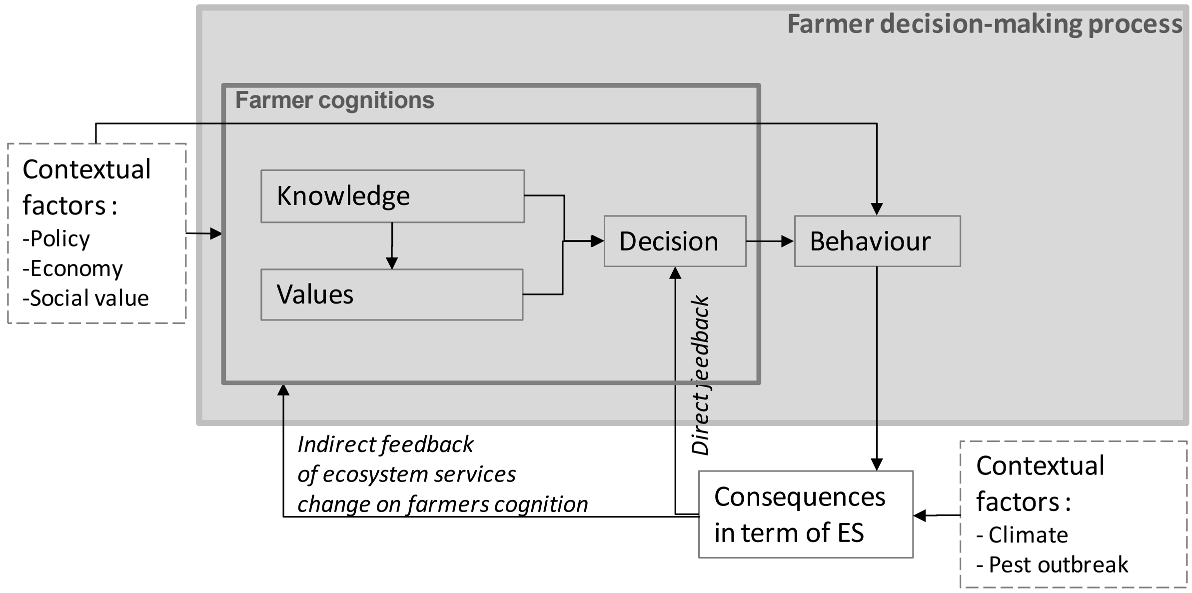


Figure 1: Socio-cognitive conceptual model of ecosystem services feedbacks on farmer behaviour.

Feedback from changes in ecosystem services supply to farmers' cognitions and behaviours can be either direct, affecting only the perceived parameters of decision, or indirect, affecting the different cognitive components underlying the behaviour (Meyfroidt, 2012).

3 Social-ecological system and methods

3.1 Study area

The study site (45°03' N, 6°24' E) is part of the Ecrins National Park in the Central French Alps, and located on the south-facing slopes (13 km²) of Villar d'Arène, a municipality of 270 inhabitants spread in three hamlets located in the lower part of the site (Figure 2). Climate is subalpine with a strong continental influence due to a rain shadow with respect to dominant westerly winds. Mean annual rainfall is 956 mm, though variable across years, (drought occurred several times over the last decade) and the mean monthly temperatures range between -4.6°C in January and 11°C in July (at 2050 m a.s.l.). Most of the upper slopes (above 2200 m) of Villar d'Arène have been extensively grazed continuously for centuries, and will be further called "Alpine meadows". Since the 20th century, the lower slopes (1650-2000 m), that were formerly terraced, ploughed and used for cropping ("terraces" henceforth), are cut for hay during summer or grazed during spring and autumn (Girel et al., 2010). Some, but not all, of these terraces are manured, requiring mechanised access. Intermediate unterraced grasslands (1800-2500 m) ("unterraced grasslands" henceforth) have been managed for hay production since the 1700s, but since the 1970's mowing has been gradually ceased over 75% of the area, which is now lightly grazed in early summer (Figure 2). Management practices are overall at low intensity, with low stocking rates, very low manure inputs (every two or three years) and a single annual hay cut. These effects of past land use (presence or absence of cultivation) and current practices (presence or absence of mowing and of manuring) have shaped the landscape into a mosaic of land management types resulting in distinct patterns of fertility, floristic and functional composition, and associated ecosystem functioning (Quétier et al., 2007a; Robson et al., 2007).

A key element of farmers strategy is fodder self-sufficiency to bridge the very long winter (6-7 months) during which stock is kept indoors, because they can not support fodder expenses. This strategy has been challenged by recent droughts, and by a vole outbreak that affected the area in 2009-10. Some farmers mow additional parcels outside of the study area and one farm ([E11] see codification below) and the majority of its parcels are located in the neighbouring municipality. The eight farmers managing the study area can be classified into three categories according to their production: (1) sheep farmers producing lambs [referred hereafter by the following codes : E10, E9, E8](mean = 21 livestock units (LU), 19 ha); (2) cattle farmers breeding calves and heifers for dairy farms [E7, E6, E11] (mean = 67 LU, 55 ha), (3) farmers raising both sheep and cattle [E2, E1](mean = 54 LU, 48 ha). During summer, most of the alpine meadows are grazed by a shepherd who manages local farmers' sheep along with his own flock (around 1400 sheep in total). The remaining alpine meadows are divided into paddocks for cattle grazing. Only two farmers sell a part of their production by direct selling. Five farmers are full-time, one is part-time farmer and two are retired but continue to farm, but usually one member of the household works outside of the farm (Deboeuf, 2009).

These farms are recognized as part of a "Less Favored Area" characterised by the combination of a short growing season (April-September) because of a high altitude (elevation ranges between 1552 and 2442 m a.s.l.) and steep slopes (from 0 to more than 50°). Hence, compensations for low productivity by European subsidies constitute a significant share of farmers' income. In addition the Ecrins National Park supports the establishment of agri-environmental measures for farmers to conserve mowing practices that maintain species and habitat biodiversity of the area.

Grasslands are collectively managed through an association called "AFP" (Association foncière pastorale) created in 1975 in which agricultural parcels of all landowners are pooled together and allocated among farmers, in order to lower constraints (ex. production costs, accessibility to parcel) and increase the average size of parcels.

In addition to agriculture, tourism is a dominant economic activity in the region, which is recognized for its aesthetic, cultural and conservation value and offers opportunities for recreation.

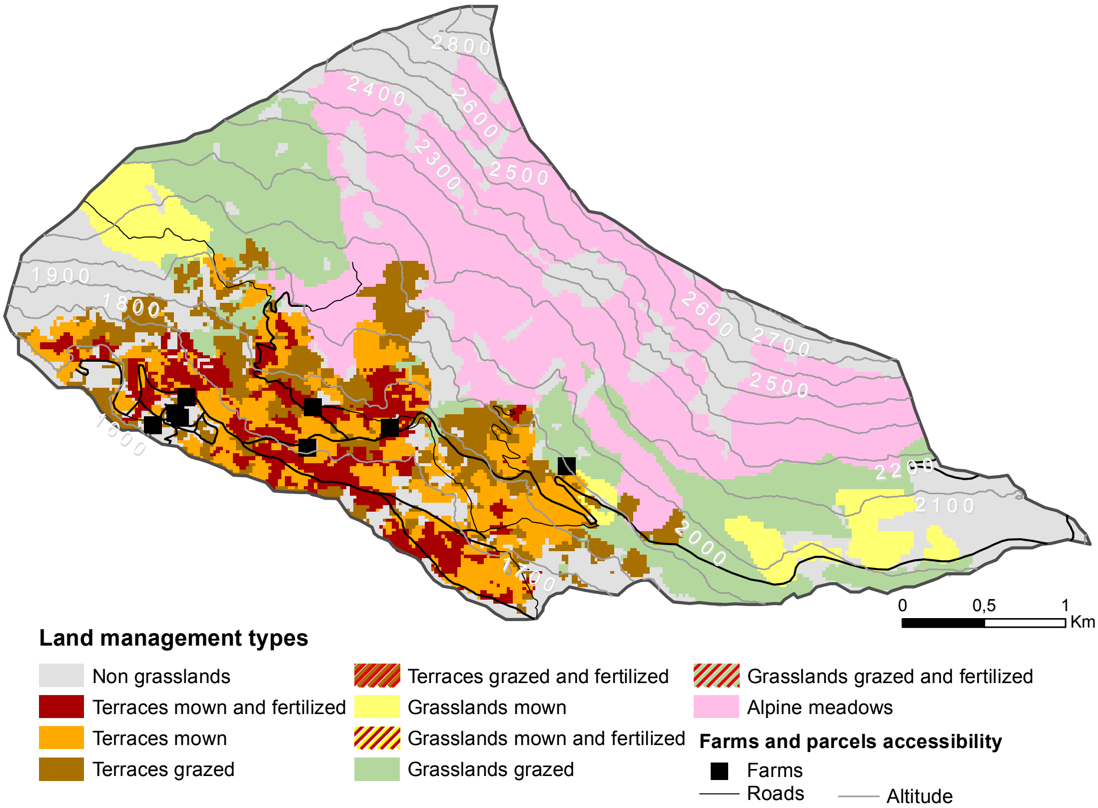


Figure 2: Grassland management types and location of farms and roads (modified from (Lavorel et al., 2011)).

3.2 Data collection

Qualitative and quantitative data used in this study were collected using a set of complementary methods: a participatory approach which took place with farmers between May 2009 and January 2012, and ecological field data that was collected between 2003 and 2011.

The main data on values, knowledge and behaviour towards ecosystem functions were obtained from actions, discussions and short questionnaires recorded during a collective role playing game (called hereafter the “feedback game”) that was conducted with seven (out of the 8) farmers of the site in January 2012. The feedback game aimed at understanding how ecosystem functions and other drivers are taken into account in land management decisions in different contexts (ecosystem functions delivery, socio-economic component and droughts). The role playing game method was chosen (i) to distinguish between what people say (‘espoused theory’) and what they do (‘theory in use’) (Jones et al., 2011), (ii) to explore if collective dynamics influence farmers’ behaviour, and (iii) to present to farmers the effects of their adaptive management in response to climate and socio-political change on ecosystem services delivery (ecosystem functions). Indeed, the feedback game was built on the outcome of a first role-playing game (called hereafter the “scenario game”) with the same farmers (April 2011) that aimed to identify and map farmers' land-management adaptations to integrated climate change scenarios (Chapter 5).

	Climate alternatives	
Drivers	**"Drastic"**	**"Intermittent"**
Season of drought and occurrence	Spring drought during four consecutive years	Spring or summer drought every two years
Effects on vegetation	Change in species composition. Development of species adapted to drought (eg. *Festuca paniculata, Carex sempervirens*)	No change
Effects on biomass production	Decrease by more than 50%	Decrease by 15% during drought years
Effects on water quantity (springs)	Decreased flow of all springs, even quenching of the less productive ones	Decreased flow of the springs
	Socio-economic alternatives	
	"Local"	**"International"**
Consumption demand	Local and high quality products	Cheapest prices
Aim of agricultural subsidies	To maintain both an agriculture with quality production and a high level of ecosystem services and biodiversity conservation. High subsidies but more restrictive in term of expected outcomes than in the "International" alternative.	To maintain open landscapes and production of environmental services such as carbon sequestration. Lower subsidies than on the local alternative, but less restrictive. A minimal income is guaranteed to farmers
Subsidies	-20% of CAP pillar 1 support: no minimum guaranteed. Agri-environmental measures (AEM) : Bonus for biodiversity with commitment to results (e.g. maintain plant diversity) : 210€/ha (maximum 10000€/farm) Strengthening of eco-conditionality requirements for funding (e.g. manure control)	-20% of CAP pillar 1 support: subsidies generally decoupled but minimum guaranteed (1 yearly minimum wage). Agri-environmental measures (AEM): Bonus for maintaining grasslands Carbon credits : 76€/ha (maximum 76000€ / farm)

Table 1: Drivers and related assumptions describing the four scenarios combining climatic and socio-economic alternatives.

The feedback game used a subset of three scenarios: (1) repeated droughts occurring every four years for four consecutive years combined with a socio-economic context assuming demand for local product and area-related agricultural subsidies (see Table 1 for more details) (further called the "drastic and local" scenario, D); (2) alternating favourable climatic years intermittent and droughts combined with a globalized socio-economic context (further called the "Intermittent and international" scenario, I); (3) repeated droughts as in D, with combined with the current socio-economic context (further called the “drastic anticipation” scenario). These three scenarios were used as an experimental design to vary the levels of ecosystem functions and other drivers and analyse their effects of on farmers behaviours.

The feedback game used the same equipments (board game representing land management type across the landscape and pieces representing cattle and fodder harvested) and rules (number of each type of pieces per land management type) as the scenario game where farmers had to feed their herd taking into account grassland productivity, based on different scenarios (for more details on the methods and results see chapter 5). The outcome of the scenario game was used to design the initial land management board of the feedback game using the adaptation taken by farmers to alternative scenarios contexts (Table 1). In the feedback game additional information beyond production was given for a set of ecosystem functions regarded as important by stakeholders in previous studies on the site (Chapter 3)(Lamarque et al., 2011b) : forage quality, forage quantity, date of flowering onset, litter quantity, plant diversity, aesthetics, water quality, nitrate leaching and carbon storage. Each session was composed of one round corresponding to a year where farmers were projected in a 2030-like situation and could consider the effects of their adaptations taken in the scenario game. Information was given on the level of ecosystem functions change between the current situation and 2030 (percentage increase or decrease) by land-management type and between practices (i.e manuring vs. non manuring and grazing vs. mowing) (Table 2). Ecosystem functions data presented to farmers (Table 2) were obtained from spatially-explicit models based on plant and microbial traits and abiotic parameters used to project expected changes in ecosystem functions in response to drought and management (Chapter 6).

The feedback game started with the individual filling of a table where each farmer ranked the value he/she attributed to each ecosystem function or service on a five levels Likert-scale (very low to very high value). This was followed by a group discussion on the attribution of values and by a detailed presentation and discussion of each service. The same day, three game sessions were conducted in order to observe and discuss farmers’ behaviour in: response to ecosystem functions change by 2030 caused by each of the three scenarios. At the end of each session, farmers were asked to write the reasons for adopting a given practice for each cell of the board game. The game finished by a general debriefing.

	Carbon storage		Nitrate leaching		Forage quantity		Litter quantity		Forage quality		Plant diversity		Flowering onset	
	D	I	D	I	D	I	D	I	D	I	D	I	D	I
Manuring vs. not manuring														
Mown terraces						↗		↘					↗	↗
Grazed terraces						↗	↗							
Mown unterraced grasslands	↘	↘					↗		↗				↗	↗
Grazed unterraced grassands														
Grazing vs. mowing														
Manured terraces						↘	↗	↗	↘		↘	↘	↘	↘
Not manured terraces					↘	↘	↗	↗	↘		↘	↘	↗	↗
Not manured unterraced grasslands			↗	↗	↗	↗	↗	↗	↘	↘	↘	↘		

Table 2: Ecosystem functions change (decrease (↘) and increase (↗) greater than 10%) between practises in each category of grassland, for the drastic and local scenario (column "D") and the intermittent and international scenario (column "I").

To verify and/or complement the information on values, knowledge and behaviour towards ecosystem functions obtained in the game, data from other sources were also combined and cross-checked. Semi-directed individual interviews were carried out with the eight farmers in summer 2009 about farm structure and features of the herd, forage resource management practices (Deboeuf, 2009). Interviews included a participatory photomapping where interviewees drew their own parcels over aerial photographs, and described the management (i.e. mowing, grazing, manuring, dates and stoking rate) as well as each parcel's 'field function', i.e. the main role assigned by farmers (Fleury et al., 1996). Field functions described by farmers for each parcel were coded according to the expectations on output: (i) both quantity and quality of fodder are expected, (ii) only quantity expected, (iii) only quality expected. We used this land management and field functions data to compare perceptions and behaviours in the field of farmers to game results.

Other sets of surveys were carried out from 2009 to 2011 with the farmers by (i) semi-directed individual interviews about knowledge and adaptations to past droughts (Nettier et al., 2010), (ii) adaptations to future climatic and socio-economic change according to four scenarios (Chapter

5), and (iii) a group interview conducted in January 2010 with 3 farmers and inhabitants to elicit their perceptions of biodiversity and ecosystem functions related to management of mountain grasslands (Chapter 3)(Lamarque et al., 2011b). We used these results to verify or complement some elements of discussion recorded during the feedback game.

Ancillary spatial data was also used to study the effects of contextual factors on land management behaviours: a current land-use map of the site constructed using a combination of cadastral (1810 to 2009) and aerial photographic data (since 1952) (Girel et al. 2010), a 10 m×10 m Digital Elevation Model and settlements, farms and road locations digitized from the 1:25000 topographic maps (IGN).

3.3 Data analysis

The different data contributed to understanding different components of the farmers' decision-making process (Figure 1). Interviews and game discussions were recorded, typed and coded by themes using Nvivo 9 to extract the different components of decisions-making (value, knowledge, decisions) for each ecosystem function. Boards of the feedback role-playing game sessions, representing management practices by each farmer for each type of grassland, were vectorized to analyse farmers' behaviour in a given context of climate and socio-economic scenario and associated effects. Maps resulting from participatory photomapping were digitized and georeferenced (with ARCGIS, ESRI), to overlay with the other maps.

Statistical analyses were carried out in R (R Development Core Team, 2008) to analyse the relationships between actual land management behaviours (mowing, grazing or manuring) and potential drivers (Table 3). This allowed observing if relations between behaviours and drivers discussed during the game process were consistent with those in the real life. Moreover, expected quality and quantity variables (field functions) were used to add spatial information to ecosystem functions values discussed during the game.

The entire feedback loop from change in ecosystem services supply to farmers' behaviours was then analyzed by combining all this data and using the « process-tracing » approach (George and Bennett, 2005) to explore individually each component of the conceptual model before considering links between them (Figure 1) (as in Meyfroidt, submitted). This method attempts to identify the causal chain and mechanisms between independent variables (cognitive factors and contextual factors) and the outcome of the dependant variable (farmers' behaviours). By tracing the processes and all the necessary implications of the main hypothesis, the alternative paths through which the outcome could have occurred were identified and tested (George and Bennett, 2005).

Variables	Description	Chi-square test	ANOVA	Linear regression A	Linear regression B	Linear regression C	Logistic regression A	Logistic regression B
Expected quality	Parcels where quality is expected by farmers. Quality only, or together with quantity	X	X					
Expected quantity	Parcels where quantity is expected by farmers. Quantity only, or together with quality	X	X					
Manuring	Presence/absence of application of manure in the parcel	X					*	
Mowing	Mowing vs. grazing practice in the parcel							*
Mowing date	Real mowing date (day) (for the year 2009)		X	*	*	*		
Date of flowering onset	Modelled date of the beginning of flowering onset of grasses (day) (Lavorel et al, 2011)			X				
Plant diversity	Modelled Simpson Index (Lavorel et al, 2011)				X			
Slope	Slope (degree)						X	X
Elevation	Log 10 of mean elevation of the parcel (m)					X	X	X
Distance to road	Log 10 of Euclidian distance from the middle of the parcel to the road or track suitable for vehicles (m)					X	X	X
Distance to farm	Log 10 of Euclidian distance from the middle of the parcel to the farm (m)					X	X	X

Table 3: Summary of the statistical analyses realized at parcel level (excluding alpine meadows). Dependent variables are depicted by "*" and independent variables are depicted by "X". ANOVA and Chi-square tests discriminate pairs of variables depicted by "X".

4 Results

This section presents successively each component of our conceptual approach (Figure 1): (1) cognitive variables (knowledge and values) about ecosystem services and practices, (2) behaviours acted and explained by farmers and (3) influence of environmental cognitions and ecosystem services on farmers land management behaviours. Finally, (4) we explore reasons other than ecosystem services which influenced farmers' behaviour. As no strong differences were observed among farmers in ecosystem functions knowledge and uses, overall results are presented hereafter and differences are specified only when relevant, using the farmer's code between square brackets. The presence of a farmers' code indicates that this farmer expressed his/her view, and the absence means an absence of view expressed by this farmer rather than a disagreement as the game discussions were collective.

4.1 3.1 Farmers' environmental cognitions

4.1.1 Knowledge

Farmers' knowledge extracted from the feedback game discussion is depicted in this section in terms of understanding and perceptions of: (1) each ecosystem functions and its relationships to others, (2) relationships between ecosystem functions and agricultural practices, (3) effects of contextual factors on ecosystem functions.

Discourse analysis elicited knowledge of farmers around ecosystem functions and identified two kinds of relationships: (1) direct links among ecosystem functions and (2) links between ecosystem functions and practices (Figure 3).

The ecosystem functions described by researchers were all known by farmers except nitrate leaching and carbon storage which required more explanations. They were aware about the existence of ecosystem services and had knowledge about them, even without calling them "ecosystem services" (Chapter 3)(Lamarque et al., 2011b; Nettier et al., 2010). Relationships mentioned differed in terms of directions of flows between services. Relationships underlined that services can mutually influence each other, or only influence the production of other services, or only be influenced by other services. Plant diversity, flowering onset and litter quantity were considered to be influenced by or to influence other services. Plant diversity was seen as positively affecting aesthetic [E11 and E6], ("A *beautiful grassland with a lot of flowers, it's more beautiful than a grassland with only Queyrelle"* (*Festuca paniculata*) [E8]), forage quality [E11, E6] ("*diversity corresponds to quality*" [E1]) and to a smaller extent flowering onset, which is also more strongly influenced by altitude [E11]. Plant diversity was considered to be itself affected negatively by litter quantity [E6, E1, E9]. Flowering onset was perceived to influence aesthetics ("*in the spring during full bloom*" [E6]). Farmers knew that litter quantity increases in unterraced grasslands colonized by one grass species, *Festuca paniculata* [E6] *which grows in tussocks with dense tall canopies*. Aesthetics, forage quantity and forage quality were not considered as influencing other services supply, but farmers thought that aesthetics was influenced positively by plant diversity [E6 and E11] and flowering onset as stated above. Forage quantity was said to be positively affected by litter quantity ("*after one year, we can see the effect during the spring on plant re-growth on grassland which are grazed only a little bit. It's* *protected from frost*" [E6 and E11]) by some farmers, or negatively by others [E11, E9 and E1] ("*in a grassland that I have not grazed a lot, in autumn and even next spring nothing regrows*" [E8]). Finally, relationships between nitrate leaching or carbon storage and other services were not mentioned, even after our explanations.

Regarding practices, manuring was not perceived to affect nitrate leaching in their agricultural system ("*We manure only with natural fertilizer (manure). It is not certified organic, but we do not use mineral fertilizer, so we are far from this kind of problem*" [E11]; "*Rain or snow seep manure into the soil. There is no leaching*" [E11]). But some farmers were careful about nitrate leaching because of legislation [E1, E9, E6, E11] ("*I take that into account because I have a plan for spreading manure agreed with the authority"* [E11]*)*. Manuring unterraced grasslands increases forage quantity [E6, E1, E9] ("*today, everything shows me that fertilizing increases forage quantity*" [E11]) and quality (E11, E9], and also plant diversity [E11, E6, E9]. But farmers considered that it is not good to manure more often or in large quantity ("*do not manure beyond some limits, because after you change the flora*"). Farmers considered that spreading manure in autumn was more efficient compared to spring (except [E6] who considered that manure is more efficient in spring when plants grow), and avoided soiling fodder ("*In the autumn manure rots better than during spring when it's dry. After we have it on fodder*"[E7]). Mowing was considered to increase plant diversity [E11] ("*farmers are all aware that the floristic diversity will change if we stop mowing. We saw it after the vole outbreak*" [E1]) but also directly aesthetics [E11, E6, E1] ("*it maintains an open landscape*" [E9]). Moreover, farmers asserted that the decision to mow was influenced by productivity of a given year since some of them do not mow when they do not consider the quantity of forage worthwhile [E11, E6, E1]. Farmers believed that mowing and especially the mowing date influenced the quality (which was said to decrease with late mowing date; [E11, E6, E9]) and quantity (which increased with late mowing date; [E11, E6, E1, E9]) of forage harvests ("*We wait as long as possible until plants are at flowering stage. We maximize quantity. But that's not the best [for quality]"* [E9]), leading to a trade-off between both services. But a late mowing date was perceived as increasing plant diversity [E1, E9] ("*Mowing too early, before July the 20th, doesn't leave plants time to set seed and then decreases the number of species*" [E6]) and then indirectly forage quality. All farmers seemed to agree that lower parcels (terraces) are mown at the beginning of July, and higher parcels not until the 10th August in years with early vegetation onset, and some years not even before the 20th August. Mowing date was in part motivated by the date of flowering onset [E11, E1]. Finally, ecosystem functions motivating grazing were not mentioned. Grazing was mentioned as having negative effects on aesthetics [E11, E6, E1] and plant diversity [E11, E1, E9], and also as decreasing litter quantity, though less than mowing [E6, E1, E9] ("*the sheep do not put their head into [Festuca paniculata], but cows manage to pull out a few*" [E11]). This representation of knowledge shows the influences of some ecosystems services on practices as well as effects of practices on the delivery of ecosystem functions.

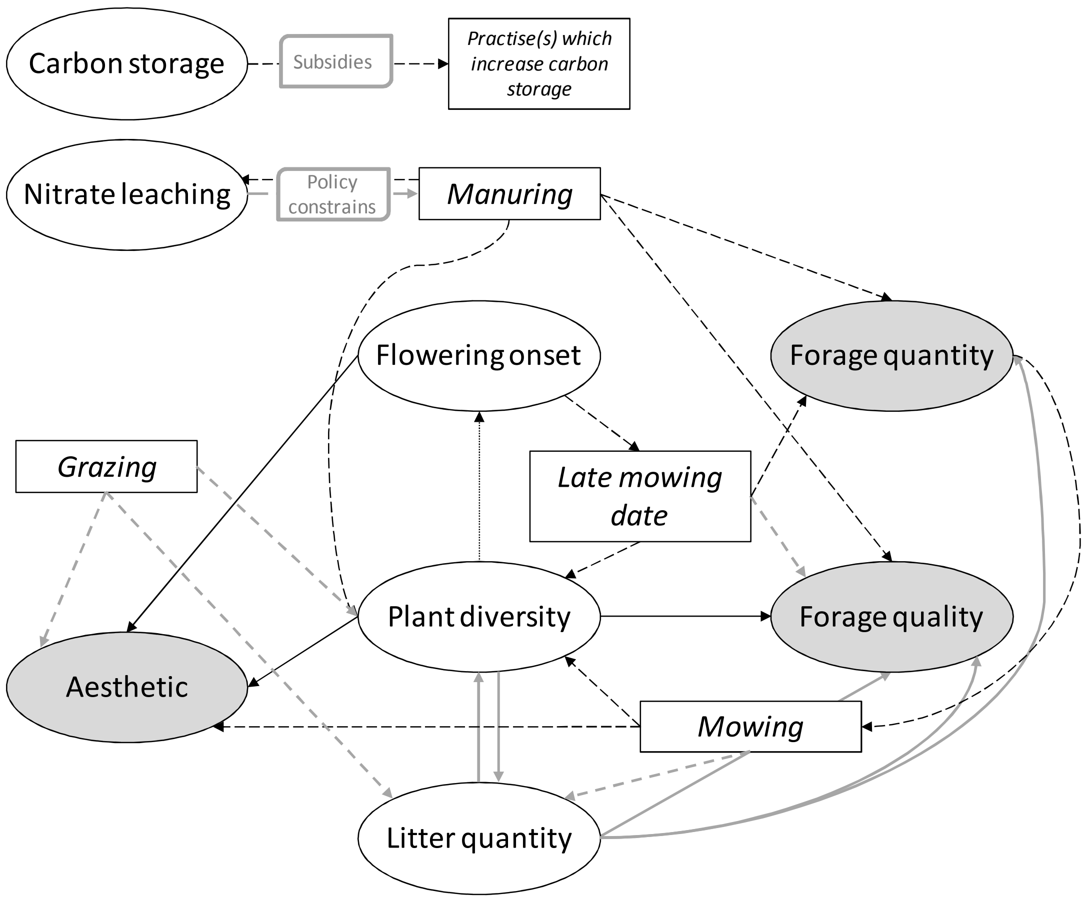


Figure 3: Conceptual representation based on farmers' discourses on values and knowledge about the relationship between ecosystem functions (ES) and land-management practises. Rectangle boxes indicate practices and ellipses indicate ES. Dashed arrows indicate links between practices and ES and plain arrows indicate links between ES. Grey arrows indicate a negative effect and black arrows a positive effect. Except for the effect of litter quantity on forage quantity, farmers agree on all the relationships. Note that ES in grey are seen as final ES by farmers while the others are considered as intermediate ES or ecosystem functions.

In addition to the knowledge described above, the effects of additional factors, such as climate, altitude or a recent vole outbreak on ecosystem functions and practices were discussed. Effects on practices will be presented in the section on "alternative hypotheses". Rainfall influenced forage quality ("*in 1988, rain occurred throughout June, there was so much fodder that we could not give it all, we wasted a lot because it was hard, coarse and the sheep didn't want to eat it*" [E9]), and forage quantity ("*summer rains lead to a bit of second growth*" [E11], "*with this spring drought we did not have a lot of forage*" [E8]). Forage quantity was also known to be influenced by temperature and altitude. "*If vegetation starts to grow too late, at 2000 meters of altitude if a cold snap occurs, the vegetation does not restart*" [E8]. Altitude was perceived as influencing the date of flowering onset more than plant diversity [E11, E1]. The presence of snow was considered to affect litter decomposition ("*snow is also needed to rot plant litter*" [E9]).

4.1.2 Values

First, we present ecosystem functions desirability in terms of their respective importance for farmers attributed through paper ranking tables filled individually. Second, we describe the reasons of these rankings expressed by farmers during a collective discussion are described.

Averaging the scores of all farmers, ecosystem functions were ranked by decreasing value as follows: Forage quality, plant diversity, forage quantity, water quality (the ecosystem service related to nitrate leaching ecosystem function), aesthetics, litter quantity, flowering onset, nitrate leaching, and carbon storage. High values were attributed to final services and low value to intermediate services except plant diversity. However, this ranking order differed among farmers (Table 4). Farmers consistently attributed high values to some services like plant diversity, forage quality and forage quantity, while there was more heterogeneity in values attributed to other services.

	Very low	Low	Medium	High	Very high
Forage quality				E6,E7	E1,E9,E10,E8,E11
Plant diversity conservation				E1,E10,E11,E6,E7	E9,E8
Forage quantity			E10,E7	E1,E9,E6	E8,E11
Water quality		E1	E11,E6,E7		E9,E10,E8
Aesthetics	E8		E1,E11	E9	E10,E6
Litter quantity	E9	E8	E1,E11	E6	E10
Flowering onset		E10,E11	E1,E9,E8	E6	
Nitrate leaching	E8	E1	E9,E10,E11	E6	
Carbon storage	E8	E9,E10	E1,E11	E6	

Table 4: Ecosystem services and functions with their values attributed by each farmer, sorted in decreasing order of value.

Forage quality was considered as highly desirable for herd welfare ("*There is a difference between fodder, and a palatable fodder consumed by cows*" [E11]) or for some part of the herd with higher needs such as lambs or dairy cows [E1], and for some farmers was complementary to forage quantity. "*That's nice to have fodder in quantity but if it's crap fodder … you have only crap fodder*"[E1]; "*it's the balance between quality and quantity that is interesting*" [E6]. But forage quality was also a factor contributing to farm economy ("*If we do not have quality fodder , we will have to buy quality fodder to compensate*" [all farmers]). This was also true for forage quantity. "*In the cost of one hectare of mowing grassland, there is also the result in terms of forage quantity and quality to take into account*" [E11]. In addition to this information on value,

Figure 4a shows the location of parcels where quality and/or quantity were expected according to the field functions mapped by farmers. Plant diversity was also highly valued by farmers either for its contribution to forage quality [E11], or for aesthetics or both [E6]. This is consistent with the indirect links suggested between plant diversity and forage quality or aesthetics (Figure 3). Medium values were assigned to litter quantity by farmers considering on one hand a positive short-term effect on vegetation re-growth due to protection against frost and a fertilizing effect of litter when mown every couple of years, and on the other hand a negative long-term effect as litter chokes out vegetation and then decreases forage quantity and quality [E11, E1]. Farmers considering only negative or positive effects attributed respectively a low [E9] or high value [E6] to litter quantity. Carbon storage was scored lowly probably due to a lack of knowledge rather than due to undesirability. Nitrate leaching was attributed a low value, probably because farmers did not feel concerned by nitrate leaching, or because it was seen as having a negative influence on water quality which was generally highly valued.

4.2 Farmers' behaviour

This section describes, by practice, first behaviours adopted by farmers during each game session based on board game analyses and second actual behaviours quantified by our statistical analyses of the farmers maps of practices in 2009. A summary of both data sets is given in Table 5. We used additional analyses at the level of parcels or grasslands types to explore spatial variation in behaviours.

4.2.1 Manured vs. unmanured

In the "drastic and local" scenario all farmers stopped to fertilize terraces and unterraced grasslands, except [E11] and [E1] who continued to fertilize some terraces. By contrast, during the second game session ("Intermittent and international scenario"), they all increased the number of terraces manured, except [E9] who stopped to manure them. Finally, in the last game session focusing on the anticipation of practices in the case of "drastic anticipation" scenario, [E11] and [E6] stopped to manure terraces, but [E1] manured them. [E11] and [E1] manured mown grasslands. [E6] manured grazed unterraced grasslands, while [E11] only fertilized some of them. [E9] did not change compared to the current situation.

In the actual practices, farmers did not manure all their land (Figure 4b and Figure 5). In 2009, they manured only some mown terraces, except one farmer [E11] who manured also mown unterraced grasslands. Grazed unterraced grasslands were not manured. Sheep farmers did not manure at all and farmers having both sheep and cattle farmers used only cattle manure.

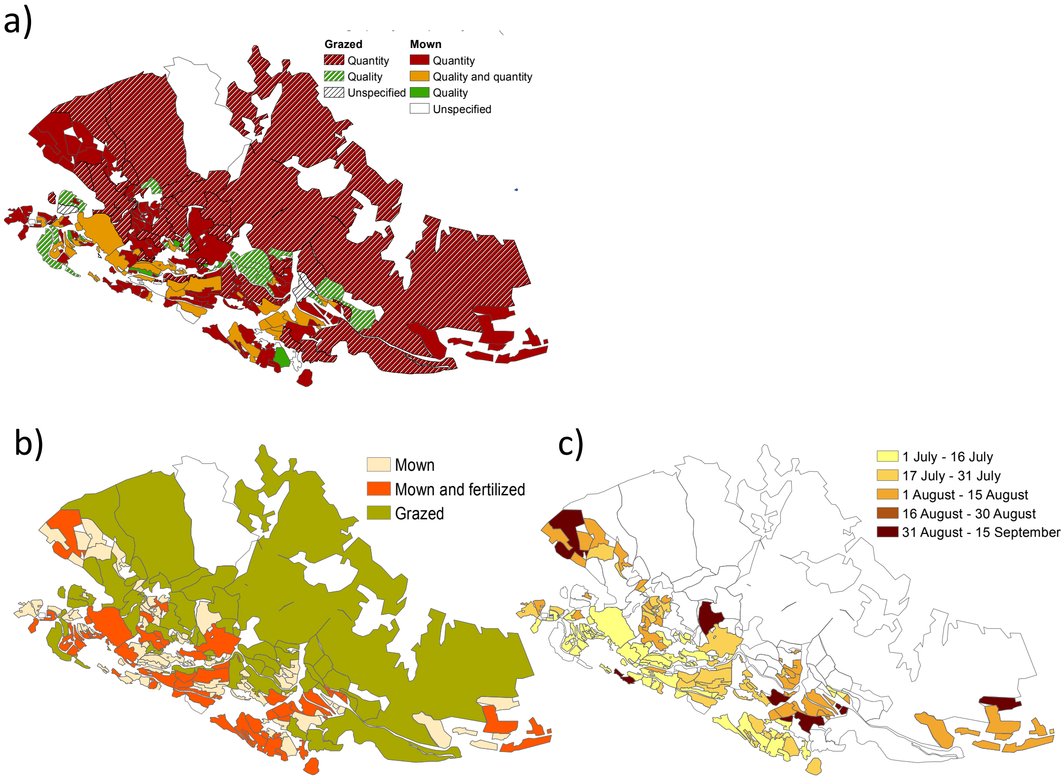


Figure 4: Maps made by farmers during the 2009 interviews: (a) farmers expectations about forage quality and quantity (colours) for mown (plain) or grazed parcels (shaded); (b) current practices and (c) actual date of mowing.

4.2.2 Mowing vs. grazing

In game session 1 ("Drastic and local"), some farmers mowed terraces [E9], but others grazed them [E11, E6, E9, E7, E10]. Unterraced grasslands were mown by some farmers [E10, E7], but other ceased their mowing [E11, E1, E6]. In game session 2, (Intermittent and international), terraces were mown, and mowing was even resumed on some grazed terraces [E11, E6]. Unterraced grasslands were mainly grazed except [E11] who continued to mow some of them. Only [E11 and E1] manured some unterraced grasslands. In game session 3, [E11] continued to mow them and [E6] even mowed and manured previously grazed unterraced grasslands.

In the actual practices, farmers organized their land management around spring grazing and mowing, because summer alpine meadows are large enough to ensure flexibility in forage resources (Figure 4b and 5). During autumn, the herd grazed very extensively on the re-growth of mown or spring grazed grassland. Areas of grazed vs. mown unterraced grasslands were adjusted to herd size ("*Our herd is our business, therefore we keep our herd and we adapt the* *rest on the herd*" [E11]), while the remaining area was used to mow, leading some farmers [E11, E6, E1] to harvest part of their fodder in other municipalities.

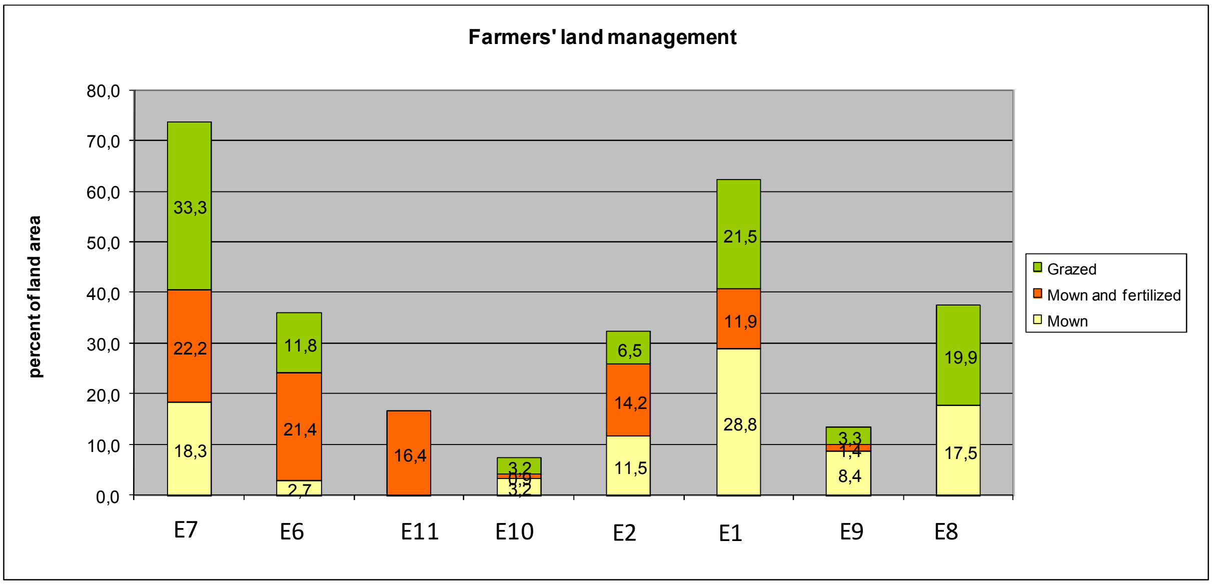


Figure 5: Percentage and area (ha) of land managed with each type of practices by farmers on the study site outside of grazing in alpine meadows, which was invariant across scenarios. Note than [E11] grazed in the neighbouring municipality of the study site, and [E11, E6 and E1] harvested only part of their fodder from the study site.

4.2.3 Date of mowing

Dates of mowing were not discussed during game sessions, but during interviews and participatory photo mapping, farmers explained that dates of mowing are spread between 1st July and mid-September (Figure 4c).

	Manuring				Mowing				Date of mowing			
	MT	**GT**	**MAG**	**GAG**	**MT**	**GT**	**MAG**	**GAG**	**MT**	**GT**	**MAG**	**GAG**
Actual practices (2009 data)	Y/N	Y/N	Y/N	Y/N	Y	N	Y	Y/N	1 July -> 15 Sept	14 July -> 15 August	5 August -> 31 August	15 August
Scenario "drastic and local"	-	Y/N	N	N	-	N	N	N	-	-	-	-
Scenario "intermittent and international"	Y	Y/N	Y/N	Y/N	Y	Y/N	Y/N	N	-	-	-	-
Scenario "drastic anticipation"	Y/N	N	Y	Y/N	Y	N	Y	Y/N	-	-	-	-

Table 5 : Farmers' behaviours in reality and in each scenario (game session) by main type of grasslands. MT : Mown terraces; GT: grazed terraces, MAG: Mown unterraced grassland, GAG: grazed unterraced grasslands. "–" means : no information. "Y" means they adopted the behaviour and "N" they didn't.

4.3 Farmers description of the influences of ecosystem functions on their behaviours

This section describes, for each practice, the influences of ecosystem functions on the behaviours adopted by farmers, according to farmers' explanations during game sessions (Table 6) or actual spatial data analysis.

	Manuring	Mowing	Date of mowing
Forage quality	X	X	
Plant diversity conservation	X	X	
Forage quantity	X		
Litter quantity		X	
Flowering onset			X
Nitrate leaching			
Carbon storage	X	X	

Table : 6 Ecosystem functions influencing farmers decisions to adopt a practice, according to farmers accounts.

4.3.1 Manured vs. unmanured

Farmers did not manure when this practice did not maintain or increase forage quantity ("We *will not manure if this does not bring quantity*" [E11]), forage quality or plant diversity (as in the drastic-local scenario), while they manured when it did (as in the intermittent-international scenario). Manuring of parcels to increase forage quality coincided with the desire to increase forage quantity both in reality and in the game sessions. One farmer manured terraces and another one unterraced grasslands to increase carbon storage and hence to receive carbon credits as proposed in the intermittent-international scenario. Nitrate leaching was never mentioned during game sessions as a reason to adapt manure practices.

The comparison between the map of actual practices (Figure 4c) and the map of expected quantity and quality (Figure 4a, field functions) showed that manure was applied mostly on parcels where quantity was expected (X^2 = 20,07, df = 1, p-value = < 0,001, n = 217) at the expense of expected quality (X^2 = 4,33 , df = 1, p-value = < 0,001 , n = 217).

4.3.2 Mowing vs. grazing

Some farmers attributed their decisions to mow terraces because of its positive effect on forage quality and decreasing effect on litter quantity [E9] (in the drastic-local scenario) or to favour plant diversity and forage quality (in the intermittent-international scenario) [E1, E7]. Date of flowering was also cited once [E1] as a factor influencing decision to mow terraces. Unterraced grasslands were also mown by some farmers to increase or maintain plant diversity and decrease litter quantity [E10, E7], in the drastic-local scenario. By contrast, these grasslands were grazed to increase carbon storage in the intermittent-international scenario [E1].

Maps of expected functions (field functionss) showed that in mown parcels farmers often expected to obtain both fodder quality and quantity (Figure 4a). Mown parcels where quality was expected were concentrated on the lower part of the slope, mixed with parcels where only quantity was required.

4.3.3 Date of mowing

One farmer indicated that, by choosing to graze unterraced grasslands to increase carbon storage and gain credits, dates of mowing on his parcels were perturbated as he had lost the possibility of later mowing in unterraced grasslands. “*I am perturbed. This grassland in altitude [unterraced grasslands] should stay mown. It’s better to have a spread in fodder than have fodder at a single altitude*” [E1]. This farmer faced a trade-off between maintaining a spread in mowing date or receiving carbon credits.

Map comparison revealed associations between early mowing (current practice) and expected quality or late mowing and expected quantity (Anova : p-value ***) (Figure 4a, 4b) . However date of mowing and actual date of flowering onset (R^2=0,01 and p-value > 0,05), or plant diversity (Simpson index) and date of mowing (R^2= 0,03, p-value > 0,05) were not significantly correlated.

4.4 Contextual factors affecting the decisions and alternative hypotheses

This section describes contextual factors (internal or external to the farm) influencing farmers decisions, using explanations by farmers (during games or interviews) and statistical analyses between spatial factors and practices (Table 7). These factors can explain why farmers’ behaviours are inconsistent with their attitudes, or in case of consistent behaviours, constitute alternatives to our main hypothesis that farmers’ land management behaviours are driven by their willingness to benefit from ecosystem functions.

4.4.1 Manuring vs. non manuring

As mowing and manuring are mechanized, constraints on the mechanisation of parcels such as slope and accessibility came out as a recurrent theme in farmers' discussions. Steep parcels can not be mechanised ("*We manure the best and flattest parcels*" [E8]. "*Manured and grazed … how is it possible? Only [E8] has flat land… anyway*" [E9]). Distance to farm was mentioned as a factor influencing manuring due to price of fuel and time to go to the parcel (except by [E11] who rents a truck bringing manure close to the most remote parcels). Other characteristics of parcels were sometimes considered [E11, E7, E1] such as proximity to dwellings, streams and water springs ("*We will not manure where there is drinking water extraction*" [E9]. "*I do not take into account distance to stream, because we have a lot of streams and with 30 meters we are far into the parcel*" [E6]. These aspects were mainly considered because of legislation and policy support including fertilisation management plans, which impose quantity, date and distance thresholds. Sheep farmers usually did not use their manure and gave it to a specialized company [E1, E9, E10, E8], which did not take liquid manure or slurry. Therefore, the capacity of slurry storage pits forced farmers to manure when it was full ("*In the lower part, I have to continue to fertilize, because I have to empty the manure pit …*" [E11]). Individual parcels were manured only once every two or three years. Finally, for some parcels, the short time between autumn grazing and snow (around 1st November on average) did not allow manuring as much as some farmers would like. Mineral fertilisation was considered as too costly to be an option.

The effects of some contextual factors mentioned by farmers were confirmed by statistical analyses on actual land use maps. The logistic regression of factors influencing manuring showed that manuring was mainly applied on gentle slopes (estimate of slope effect = -0,165 ***, R^2 = 0,06), but distance to farm, distance to road and altitude were not significant. In addition, we estimated the total area which could be manured according to the amount of manure produced depending on farm characteristics. This theoretical calculation considered farm herd size, an average annual production of manure of 4,5 T per livestock unit, a theoretical average of 15 T/ha of manure per spreading and a frequency of manuring every two or three years for each parcel. The results suggest that all farmers could potentially fertilize almost all their mown grassland at the selected frequency, except [E6] (only 50%) and [E7 and E9] (around 80%), or that they increase frequency on those parcels that are usually fertilized.

4.4.2 Mowing vs. grazing

Farmers also mentioned that mechanisation of parcels was an important determinant of conversion from mowing to grazing ("*In this parcel we can not load the hay. We need to bring it down to the road*" [E7]). At the time of data collection farmers considered that they were mowing all the mechanisable parcels (except [E11] who grazed some flat parcels in another municipality). "*Here, mowing currently grazed parcels is not possible. There is no lands were a* *return to mowing is possible*" [E1]. Mowing equipment adapted to mountain exists, but farmers asserted that it was too expensive for them. Factors related to farms' economy such as cost of mowing considering investment into equipment, depreciation and maintenance appeared often in farmers' discourses. This was even more prominent when the farmer had acquired new equipments to meet the norms of the European Union and/or when farm debt level was important. When asked to rate the importance of equipment investments compared to ecosystem functions in their behaviour, farmers responded "very high" for cost of mowing and "high" for ecosystem services (except [E6] who rated services at "medium" level). Some equipment such as manure spreader was shared between farmers to decrease costs, but each farmer usually had their own cutting equipment because they all needed it at the same time (except [E10] who did not own any equipment and borrowed it from [E9]). Agri-environmental measures provided subsidies to mow unterraced grasslands and a possibility of extra subsidy to mow less mechanisable parcels with a pedestrian mower. While this kind of subsidies substantially contributed to the farms' economy, farmers argued several times during the game that, although policy supports were carefully taken into consideration (balancing the financial amount and constraints), their amount should be higher to better compensate the actual cost of maintaining mowing practices.

Farmers discussed different elements favouring conversion from mowing to grazing. They explained that parcels surrounded by grazed parcels belonging to other farmers could be more prone to conversion, to avoid risks like trampling by cattle or fence removal. The altitude of the site did not allow multiple uses of parcels throughout the season because vegetation re-grew only at the end of summer, and in small quantities. Parcels close to the farm were needed to turn out the herd to grass during the first weeks. Grazing also required the presence of water or the possibility to install a trough. "The *problem with grazing is that we need water supply. A cow consumes 40 litres per day and there is not always an access to carry water*" [E1]. Opportunities to mow elsewhere or to acquire parcels from future retired farmers might arise, allowing remaining farms to increase their land and then fodder production, or to restrict mowing to the more mechanisable parcels and to graze the others. By consolidating land among farmers, the "Association foncière pastorale" has allowed them to increase the average size of their parcels and then to decrease their production costs, but also to have the opportunity to manage parcels they couldn't manage formerly due to conflicts between families. This could in turn favour mowing because some farmers perceived a social pressure to properly manage land, and thus to prevent shrub encroachment, especially in terraces which have a high cultural value (and suffer from cattle trampling). This is directly linked with farmers' perceptions of the social value of farming and of social pressure from landowners, other farmers and/or inhabitants. "*We will try to continue to mow as far as we can by respect towards elderly people … but on mechanisable parcels*", "*To respect their work, the terraces they built*" [E6]. "*We have to respect land. Not entering when it's wet, and not grazing when mowing is possible. When a terrace is grazed it's* *due to an accessibility issue*" [E7] "*Grazing instead of mowing is another system, the agreement of the landowner would be needed*" [E1]. Mowing also appeared as an important aspect of the farming profession for farmers, as shown in their discussions where the possibility of completely stopping mowing was always source of laughter ("*What we will do during summer if we stop mowing ? We will have a lot of time. We are not shepherds"*.)

The logistic regression on mapped data (R^2 = 0,409) confirmed the farmers' explanations that mowing, in contrast to grazing, was preferentially located on parcels with gentler slopes (estimate of slope effect for grazing = 0.43***) and further from farms (estimate of distance effect for mowing = -1.79***). Altitude and distance to roads were not significant in the model.

4.4.3 Early vs. late mowing

For farmers having contracted agri-environmental measures, this support specified dates of mowing after the 1st July on unterraced grasslands. This subsidy was perceived as a constraint by some farmers, depending on the variability in flowering date across years. Having parcels spread across the landscape increased the time taken to mow them all (although this had decreased since the re-parcelling between farmers), but the altitudinal difference between parcels allowed them to stagger the mowing over the summer season. This was seen by some of them as an opportunity [E1], while others perceived it more as a constraint and argued that ideally it would be preferable to have the entire mown area around their farms [E9, E6] ("*If land becomes available, I will stop to mow over the entire landscape and do it near my farm, to waste as little time as possible. Even if I will need to increase by two the hours per day or to take an additional worker during a few days"* [E11]). All farmers worked alone on their farm, and additional labour was hired exclusively from family when needed, e.g. for mowing [E1, E7, E9]. None of the farmers hired workers. On parcels surrounded by parcels belonging to others, farmers had to postpone mowing because surrounding parcels had to be mown first. Around the villages and campsites, some parcels were mown earlier to avoid trampling by tourists. Finally, the mowing date on a given year depended on rain to allow harvesting fodder in dry weather.

According to spatial data, although early mowing date occurred mainly on the lower part of the area and late mowing date on the upper part, some parcels broke with this rule (Figure 4d). A linear regression model confirmed that not only parcel elevation but also distance to farm had an influence on the date of mowing (Log 10 of mean elevation of the parcel: Estimate = 257,114 ***, Log 10 of mean distance to farm: Estimate = 12.87 ***, R^2 = 0,34 ***).

	Manuring	Mowing	Late mowing
Steep slope	-	-	
Altitude			+
Proximity to farm	+	-	-
Low accessibility		-	+
Proximity of dwellings or streams	-	-	
Parcels surroundings	-	-	-/+
Availability in manure	+		
Equipment costs	-	-	
Social conflicts		+	
Social value of farming		+	+
Subsidies amount		+	+
Policy or legislation constraints	-	-	+
Availability of land		+	
Snow or rain	-		+

Table 7: Summary of factors affecting positively (+) or negatively (–) the decision to adopt a behaviour (manuring, mowing, late mowing)

5 Discussion

This section first discusses the results in relation to our research question (how do farmers take into account ecosystem services in their behaviours?), looking at how ecosystem functions knowledge and values influence behaviour and how contextual factors can change their cognition or decisions. Then implications of our findings for future studies on ecosystem services are examined. Finally, some policy recommendations are discussed.

5.1 Role of ecosystem services and goods in farmers' decision-making process

At the end of the game, farmers explained that their vision of ecosystem services was different from the scientific vision. "*For us they are neither numbers nor upward or downward trends*". Ecosystem services are part of a more complex system explored in this study which looks at how farmers consider ecosystem services in their decisions. This section describes the correspondence between behaviours expected from farmers according to their attitudes (values and knowledge) towards ecosystem services, and farmers' behaviour in the game or actual life. Returning to the conceptual chain (Figure 1), three configurations emerge, explaining whether ecosystem services were taken into account in farmers decisions or not. First, some ecosystem functions were not part of farmers' knowledge or far away from their interest and therefore had low values. This was the case for carbon storage and nitrate leaching, which were thus not expected to be considered by farmers in their decisions. Yet, institutional mechanisms may lead farmers to consider these services (Vatn, 2010), as demonstrated in the second game session where contractual carbon credits could be allocated to farmers, which indeed modified some decisions. Second, some ecosystem functions were part of farmers' knowledge but attributed a low value. Regarding the date of flowering onset, farmers perceived an influence of mowing date but they did not consider themselves capable of obtaining a desirable ecosystem service delivery by their actions. To be consistent with their attitudes towards forage quality and quantity, farmers should try to adapt mowing date to flowering onset, but results showed that farmers didn't do it for other reasons (section 3.4, e.g. distance to farm, surroundings of the parcels, weather). Third, in some cases farmers had knowledge on ecosystem functions and gave them a high value. Game sessions showed that more parcels were manured when it enabled increased forage quantity and secondarily forage quality or plant diversity. According to the positive links between forage quantity, plant diversity and late mowing date, farmers could be expected to mow late. But, because late mowing decreased forage quality directly, even though it could increase it indirectly through higher plant diversity, trade-offs had to be considered. Therefore, the respective values attributed to forage quantity and quality led farmers to opt for behaviour in favour of the most highly valued ES. Whatever the behaviour adopted, in this case, farmers took multiple ecosystem services into account in their decision. These results suggest that knowledge and/or values were necessary but not decisive in farmers decisions.

Results also suggest that both direct and indirect ecosystem services feedbacks (Figure 1) explain how ecosystem services were taken into account in farmers decisions. Most changes in ecosystem functions during the game have direct feedback effects on farmers' decisions because farmers face these changes frequently (e.g. change in fodder quality or quantity due to weather conditions). But an indirect feedback was also observed with the case of carbon storage. Farmers were not aware of carbon storage before the game, and changed their values and knowledge about it, so much so that for some this factor entered in their decisions. Another example regards their knowledge about the effects of manuring on forage quantity during drastic drought. While in the previous scenario game analysing farmers' adaptation to climate change, farmers increased manuring to face droughts (Chapter 5), in the first session of the feedback game some farmers considered our results on ecosystem functions impacts (Table 2) and stopped to manure because of the inefficiency of this practice under drought. Occurrence and amount of change in ecosystem functions could also influence the feedback type. Short-term or small changes in ecosystem functions affected farmers' behaviours more through direct feedbacks leading to tactical adaptation (e.g. conversion of mowing to grazing on a given year). By contrast, greater or frequent changes in ecosystem services supply (e.g. repeated drought decreasing forage quantity, or a vole outbreak during several years) could lead to change in values. "*During years of crisis, we look first at quantity and quality, before considering colours of flowers and all these things. If you asked us the same question some years ago, we would probably not have answered the same thing*" [E11]. "*Some years ago, I was more or less independent for fodder. I was looking mainly for quality to have a specific fodder for lambs and calves*" [E1]. "*Due to the vole outbreak, we had bad fodder because soil was collected along with fodder. This led us to think differently*" [E11].

Nevertheless, it would be naive to consider that ecosystem services fully drive farmers' behaviours. Indeed, behaviours did not always correspond to their attitudes regarding ES and this could depend on the parcels considered (section 3.3). Other studies have shown that the following factors generally influence land use practices in European mountain systems: parcel characteristics (e.g. topography, location, size, land-locked position, proximity of water supply), market prices (e.g. input prices and output prices of production), policy support (e.g. types and amount of subsidies), climate (e.g. drought, frost, rain) and pest outbreaks (e.g. voles, grasshoppers) (Mottet et al., 2006; Nettier et al., 2011; von Glasenapp and Thornton, 2011). Social factors are also identified including structure of the farm business (e.g. farm type, farm size, farm economy), farmer characteristics (e.g. age, gender, education, and personality), household characteristics (e.g. level of pluri-activity, work pattern of the spouse), and structure of the social environment (e.g. local culture, social capital, information flows) (Edwards-Jones, 2006). On the one hand, our results confirmed that, although the influence of ecosystem services is not negligible, in some cases these other factors outweigh ecosystem services in farmers' decisions. For example, the difference in manuring between game sessions 1 and 2 confirmed that forage quantity influenced farmers' decisions, because they fertilised only when it was efficient. Nevertheless, some parcels were not manured (in game session 2 or reality) because some of them were not mechanisable, too far from the farm resulting in high transport cost, or at unauthorised distance from streams or settlements imposed by regulations. By contrast, in game session 1 despite the inefficiency of manuring some farmers still manured some parcels because they had to use their manure. During the feedback game farmers chose to mow some parcels to increase forage quantity and/or quality, plant diversity or decrease litter quantity but sometimes grazing was preferred due to the cost of mowing, accessibility and possibility to mechanise. Financial incentives also had importance in farmers' decision as demonstrated by the case of a farmer grazing a parcel to receive carbon storage subsidies although mowing would have been preferred for forage quality. Our results show that farmers took into account date of mowing to increase forage quantity or quality, but some parcels around villages were mown earlier than expected to avoid trampling. Conversely unterraced grasslands were mown too late because of the time needed to mow all the other parcels nearest to the farm. On the other hand, even when behaviours were found to be consistent with attitudes towards ecosystem services, alternative reasons could also have driven farmers to adopt these behaviours. Ecosystem services would thus be only one factor among others contributing to farmers' decisions. For example, some farmers mowed parcels to increase plant diversity, but this could also have been favoured by financial support from agri-environmental measures and social value attributed to mowing as part of the farming profession. Agri-environmental measures could also favour late mowing, as they impose a date threshold.

The results presented are valid for our study located in a high mountain farmer community of the French Alps, where agriculture is very extensive. Although to our knowledge no previous study has analysed the complete feedback loop, some studies suggest that farmers have good knowledge about the relationship between their practises and the functioning of their agro-ecosystem (Nettier et al., 2011; von Glasenapp and Thornton, 2011). Therefore, our results might hold for other mountain agricultural social-ecological systems because famers opportunities (policy support, economy) and constraints (e.g. topography, weather conditions, higher cost of productions) to adopt a behaviour are to a large extent likely to be similar (Mottet et al., 2006; von Glasenapp and Thornton, 2011).

5.2 Implications for ecosystem services research

The results point out the importance of considering stakeholders' perception and use of ecosystems rather than focusing only on ecosystem services supply (ecosystem functions) (Termorshuizen and Opdam, 2009). Stakeholders' perceptions and ecosystem functions sometime differ (Chapter 3) (Lamarque et al., 2011a) and the latter may not coincide with stakeholders' needs. Nevertheless, most of the ecosystem functions presented to farmers were indeed considered by them as ecosystem services (except for water quality/nitrate leaching and carbon storage of which they were not aware). Studying systemic representation of ecosystem functions by farmers (Figure 1) allowed us to also show that some ecosystem functions benefit farmers (i) either individually (forage quality, forage quantity, litter quantity, date of flowering onset, plant diversity and aesthetics) or combined together with tradeoffs (forage quantity and quality), and (ii) directly (called final ecosystem services: forage quantity, forage quality, aesthetics) and/or indirectly (called intermediate ecosystem services : flowering onset, plant diversity, litter quantity). For example, plant diversity does not seem to be considered by farmers for its intrinsic value but rather for its value to contribute to forage quality. Quality and quantity of "green" forage (vegetation) were seen as services deriving directly from the ecosystem, while quality and quantity of "dry" fodder (harvested) were considered as goods (or benefits), acknowledging that manufactured and/or human capital (e.g. farmer know-how) are required to generate a valued good from ecosystem services (for more details see chapter 2, Lamarque et al., 2011a or Mace et al, 2011). ("*The quality of fodder is linked to farmer's work. The way the grassland is managed every day*" [E11]). Moreover, our results showed the importance to not only consider perception or valuation of ecosystem services by stakeholders but also their effective uses. Ecosystem services potentially supplied (ecosystem functions), ecosystem services perceptions and ecosystem services actually used can differ according to individuals and to the spatial and temporal contexts. For example farmers might not manage a parcel with high plant diversity towards quality fodder because of topography limiting access, or because of the context forcing them to maximise quantity at the expense of quality. For the purpose of ecosystem services conservation, it is also important to consider farmers' awareness, willingness and/or ability to adopt a practice maintaining or enhancing ecosystem services delivery in the social-ecological system as a whole.

Although all components of agro-systems cannot be translated in term of ecosystem services, research focusing on ecosystem services can complement agronomic studies for several reasons. Firstly, the ecosystem services framework brings a common language easily understandable by researchers from different disciplines as well as farmers, stakeholders or policy makers (Barnaud et al., 2011). This study showed that the concept was rapidly understood, even by farmers who had never heard the term before (Chapter 3) (Lamarque et al., 2011b). Secondly, it emphasizes human-environment interactions which have been generally overlooked in other researches, and translates ecological complexity into common language, increasing the awareness of the dependence of society on biodiversity and ecosystems (Vihervaara et al., 2010). Thirdly, it allows identifying and arbitrating trade-offs and priorities at the farm, municipality or even larger scales involving beneficiaries having different interests. In this study, the national park tries to maintain mowing in unterraced grasslands to conserve this rare agro-ecosystem and related biodiversity, while farmers are interested to maximize other ecosystem services and pushed to stop mowing by other contextual factors including profitability or topography. Therefore, the ecosystem services framework could help design public policies to reconcile the interests of different stakeholders.

5.3 Policy implications

As suggested by (Prager and Freese, 2009), the degree of flexibility of management options can be a key factor for policies to be successful depending on the individual, the farm and the practice to be adopted. This tends to make agri-environmental policies based on results (e.g. a given number of determined species in the parcels) rather than on actions (e.g. requiring mowing) (Gibbons et al., 2011). This paper does not aim at comparing farmers' knowledge with scientific knowledge to verify how farmers could effectively increase or maintain ecosystem services supply by adapting their behaviours. Therefore, we are not able to say whether encouraging results alone is sufficient or if agri-environmental subsidies should also support management practices to reach results by giving more information such as technical advices. This is important because (Nettier et al., 2011) demonstrated for agri-environmental measures that farmers do not engage parcels where they are not sure of reaching the required result.

Our results highlight the importance of the level of subsidies in farmers' decisions. For example, substantial amount of subsidies for carbon storage led some of them to adopt practices in contradiction with their attitude towards other services or values. This was already observed in another study, where payments for ecosystem services encouraged people to adopt behaviours which undermine their moral sentiments for conservation (Vatn, 2010). Conversely, subsidies can also help a farmer keep or adopt a behaviour favouring some ecosystem services, if the amount adequately makes up for the loss of earnings due to more expensive equipment, fuel or time and/or if the behaviour is not restrictive for their extensive livestock rearing (Allaire et al., 2009). For example, too small subsidies to sustain mowing could lead some farmers to prefer practices that are less costly in time, financial or equipment terms. The bundle of policies offered to farmers, aiming at maintaining or increasing specific services, should avoid to create or reinforce trade-offs between ecosystem services and/or in farmers' decisions (Millennium Ecosystem Assessment, 2005; Swallow et al., 2009). For example, in unterraced grasslands, agri-environmental measures proposed by the national park to favour plant and habitat diversity pay farmers for mowing, while a carbon storage policy could support grazing on the same lands. If the latter would give higher amount, farmers could choose to contract for carbon storage, despite the long-term negative effects of grazing on other services they value.

Conclusion

To our knowledge this is the first ecosystem services study exploring the feedback between ecosystem services and farmers' behaviour through the farmers' decisions making process. By demonstrating the causal chain and mechanisms leading farmers to adopt a behaviour, our study suggests that contrary to our hypothesis land management behaviour is not always driven by farmer's willingness to benefit from ecosystem services. We obtained convincing arguments showing that farmers take into account some ecosystem services in their decisions, but we also showed that ecosystem services constitute necessary but not sufficient conditions in influencing behaviours because other key factors were taken into account. Such an approach should be tested at other sites with a greater set of ecosystem services and/or other beneficiaries and land managers.

Acknowledgments

We thank all participants of interviews, workshops and RPGs for their contributions and the time they devoted to our study. We also thank Fabien Quétier, Aloïs Artaux, Eric Deboeuf and Coline Byczek for their help during the RPG sessions, workshop and/or interviews, and the Joseph Fourier Alpine Station for logistical support. This research was conducted on the long term research site “Zone Atelier Alpes”, a member of the ILTER-Europe network (ZAA publication n°XXXXX). This work was supported by ANR through the FP6 BiodivERsA Eranet VITAL project and by the French Ministry responsible for environment (MEEDDM) through the GICC-2 SECALP project.

References

Collins, S. L., Carpenter, S. R., Swinton, S. M., Orenstein, D. E., Childers, D. L., Gragson, T. L., Grimm, N. B., Grove, M., Harlan, S. L., Kaye, J. P., Knapp, A. K., Kofinas, G. P., Magnuson, J. J., McDowell, W. H., Melack, J. M., Ogden, L. A., Robertson, G. P., Smith, M. D., Whitmer, A. C., 2011, An integrated conceptual framework for long-term social-ecological research, *Frontiers in Ecology and the Environment* **9**(6)**:**351-357.

Deboeuf, E., 2009, Adaptabilité des systèmes d'élevage de haute-montagne à des aléas. Le cas de Villar d'Arène, Enita de Clermont-Ferrand, France, pp. 91.

Diaz, S., Quetier, F., Caceres, D. M., Trainor, S. F., Perez-Harguindeguy, N., Bret-Harte, M. S., Finegan, B., Pena-Claros, M., Poorter, L., 2011, Linking functional diversity and social actor strategies in a framework for interdisciplinary analysis of nature's benefits to society, *Proceedings of the National Academy of Sciences of the United States of America* **108**(3)**:**895-902.

Fleury, P., Dubeuf, B., Jeannin, B., 1996, Forage management in dairy farms: A methodological approach, *Agricultural Systems* **52**(2-3)**:**199-212.

Girel, J., Quétier, F., Bignon, A., Aubert, S., 2010, Histoire de l'agriculture en Oisans. Hautes Romanche et pays faranchin. Villar d'Arène, Hautes-Alpes, in: *La Galerie de l'Alpe*, Station Alpine Joseph Fourier, Grenoble, France, pp. 79.

Grothmann, T., Patt, A., 2005, Adaptive capacity and human cognition: The process of individual adaptation to climate change, *Global Environmental Change-Human and Policy Dimensions* **15**(3)**:**199-213.

Kumar, M., Kumar, P., 2008, Valuation of the ecosystem services: A psycho-cultural perspective, *Ecological Economics* **64**(4)**:**808-819.

Liu, J. G., Dietz, T., Carpenter, S. R., Alberti, M., Folke, C., Moran, E., Pell, A. N., Deadman, P., Kratz, T., Lubchenco, J., Ostrom, E., Ouyang, Z., Provencher, W., Redman, C. L., Schneider, S. H., Taylor, W. W., 2007, Complexity of coupled human and natural systems, *Science* **317**(5844)**:**1513-1516.

Termorshuizen, J. W., Opdam, P., 2009, Landscape services as a bridge between landscape ecology and sustainable development *Landscape Ecology* **Volume 24**(Number 8)**:**1037-1052.

Allaire, G., Cahuzac, E., Simioni, M., 2009, Contractualisation et diffusion spatiale des mesures agro-environnementales herbagères, *Revue d'Etudes en Agriculture et Environnement* **90**(1)**:**23-50.

Barnaud, C., Antona, M., Marzin, J., 2011, Vers une mise en débat des incertitudes associées à la notion de service écosystémique, *Vertigo* **11**(1).

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R. V., Paruelo, J., Raskin, R. G., Sutton, P., van den Belt, M., 1997, The value of the worl's ecosystem services and natural capital, *Nature* **387:**253-260.

Dietz, T., Fitzgerald, A., Shwom, R., 2005, Environmental values, *Annu. Rev. Environ. Resour.* **30:**335-372.

Duguma, L. A., Hager, H., 2011, Farmers' assessment of the social and ecological values of land uses in central Highland Ethiopia, *Environmental management* **47:**969-982.

Edwards-Jones, G., 2006, Modelling farmer decision-making: concepts, progress and challenges, *Animal Science* **82**(06)**:**783-790.

Engel, S., Pagiola, S., Wunder, S., 2008, Designing payments for environmental services in theory and practice: An overview of the issues, *Ecological economics* **65**(4)**:**663-674.

Feola, G., Binder, C. R., 2010, Towards an improved understanding of farmers' behaviour: The integrative agent-centred (IAC) framework, *Ecological Economics* **69**(12)**:**2323-2333.

Foley, J. A., DeFries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., Chapin, F. S., Coe, M. T., Daily, G. C., Gibbs, H. K., Helkowski, J. H., Holloway, T., Howard, E. A., Kucharik, C. J., Monfreda, C., Patz, J. A., Prentice, I. C., Ramankutty, N., Snyder, P. K., 2005, Global Consequences of Land Use, *Science* **309**(5734)**:**570-574.

George, A. L., Bennett, A., 2005, Case studies and theory development in the social sciences, The MIT Press, Cambridge, Massachusetts.

Gibbons, J. M., Nicholson, E., Milner-Gulland, E. J., Jones, J. P. G., 2011, Should payments for biodiversity conservation be based on action or results?, *Journal of Applied Ecology* **48**(5)**:**1218-1226.

Hein, L., van Koppen, K., de Groot, R. S., van Ierland, E. C., 2006, Spatial scales, stakeholders and the valuation of ecosystem services, *Ecological Economics* **57**(2)**:**209-228.

Jones, N. A., Ross, H., Lynam, T., Perez, P., Leitch, A., 2011, Mental Models: An Interdisciplinary Synthesis of Theory and Methods, *Ecology and Society* **16**(1).

Lamarque, P., Quetier, F., Lavorel, S., 2011a, The diversity of the ecosystem services concept and its implications for their assessment and management, *Comptes Rendus Biologies* **334**(5-6)**:**441-449.

Lamarque, P., Tappeiner, U., Turner, C., Steinbacher, M., Bardgett, R. D., Szukics, U., Schermer, M., Lavorel, S., 2011b, Stakeholder perceptions of grassland ecosystem services in relation to knowledge on soil fertility and biodiversity, *Regional Environmental Change* **11**(4)**:**791-804.

Lauer, M., Aswani, S., 2010, Indigenous knowledge and long-term ecological change: detection, interpretation, and responses to changing ecological conditions in pacific Island Communities, *Environmental management* **45**(5)**:**985-997.

Lavorel, S., Grigulis, K., Lamarque, P., Colace, M.-P., Garden, D., Girel, J., Pellet, G., Douzet, R., 2011, Using plant functional traits to understand the landscape distribution of multiple ecosystem services, *Journal of Ecology* **99**(1)**:**135-147.

Lewan, L., Soderqvist, T., 2002, Knowledge and recognition of ecosystem services among the general public in a drainage basin in Scania, Southern Sweden, *Ecological Economics* **42**(3)**:**459-467.

Mace, G. M., Bateman, I., (ed.), 2011, Conceptual Framework and Methodology. In: The UK National Ecosystem Assessment Technical Report. UK National Ecosystem Assessment, UNEP-WCMC, Cambridge.

MEA, 2005, Millennium Ecosystem Assessment. Ecosystems and Human Well-being: Synthesis, Island Press, Washington DC U.S.A.

Meyfroidt, P., 2012, Environmental cognitions, land change, and social-ecological feedbacks: an overview, *Journal of Land Use Science*.

Millennium Ecosystem Assessment, 2005, Ecosystems and human well-being: scenarios, Island Press, Washington D.C., USA.

Mottet, A., Ladet, S., Coque, N., Gibon, A., 2006, Agricultural land-use change and its drivers in mountain landscapes: A case study in the Pyrenees, *Agriculture Ecosystems & Environment* **114**(2-4)**:**296-310.

Nettier, B., Dobremez, L., Coussy, J. L., Romagny, T., 2010, Attitudes of livestock farmers and sensitivity of livestock farming systems to drought conditions in the French Alps, *Revue De Geographie Alpine-Journal of Alpine Research* **98**(1)**:**383-400.

Nettier, B., Dobremez, L., Seres, C., Pauthenet, Y., Orsini, M., Kosmala, L., Fleury, P., 2011, Biodiversity conservation by livestock farmers: advantages and shortcomings of the agri-environmental scheme 'Prairies fleuries', *Fourrages* (208)**:**283-292.

O'Farrell, P. J., Donaldson, J. S., Hoffman, M. T., 2007, The influence of ecosystem goods and services on livestock management practices on the Bokkeveld plateau, South Africa, *Agriculture, Ecosystems & Environment* **122**(3)**:**312-324.

Prager, K., Freese, J., 2009, Stakeholder involvement in agri-environmental policy making - Learning from a local- and a state-level approach in Germany, *Journal of Environmental Management* **90**(2)**:**1154-1167.

Quétier, 2006, Vulnérabilité des écosystèmes semi-naturels européens aux changements d'utilisations des terres, in: *Biologie des systèmes intégrés, Agronomie-Environnement*, Ecole supérieure Agronomique de Montpellier, Montpellier, pp. 269.

Quétier, F., Lavorel, S., Thuillier, W., Davies, I., 2007, Plant-trait-based modelling assessment of ecosystem services sensitivity to land-use change, *Ecological Applications* **17**(8)**:**2377-2386.

R Development Core Team, 2008, R: a language and envrionment for statistical computing (R Foundations for Statistical Computing, Vienna).

Robson, T. M., Lavorel, S., Clement, J.-C., Roux, X. L., 2007, Neglect of mowing and manuring leads to slower nitrogen cycling in subalpine grasslands, *Soil Biology and Biochemistry* **39**(4)**:**930-941.

St John, F. A. V., Edwards-Jones, G., Jones, J. P. G., 2011, Conservation and human behaviour: lessons from social psychology, *Wildlife Research* **37**(8)**:**658-667.

Swallow, B. M., Sang, J. K., Nyabenge, M., Bundotich, D. K., Duraiappah, A. K., Yatich, T. B., 2009, Tradeoffs, synergies and traps among ecosystem services in the Lake Victoria basin of East Africa, *Environmental Science & Policy* **12**(4)**:**504-519.

TEEB, 2009, The Economics of Ecosystems and Biodiversity for Policy Makers Summary: Responding to the Value of Nature, Available at: www.teebweb.org, accessed 10 june 2010.

Tengö, M., Belfrage, K., 2004, Local management practices for dealing with change and uncertainty: a cross-scale comparison of cases in Sweden and Tanzania, *Ecology and Society* **9**(4).

Vatn, A., 2010, An institutional analysis of payments for environmental services, *Ecological Economics* **69**(6)**:**1245-1252.

Vignola, R., Koellner, T., Scholz, R. W., McDaniels, T. L., 2010, Decision-making by farmers regarding ecosystem services: Factors affecting soil conservation efforts in Costa Rica, *Land Use Policy* **27**(4)**:**1132-1142.

Vihervaara, P., Rönkä, M., Walls, M., 2010, Trends in Ecosystem Service Research: Early Steps and Current Drivers, *Ambio* **39**(4)**:**p. 314-324.

von Glasenapp, M., Thornton, T., 2011, Traditional Ecological Knowledge of Swiss Alpine Farmers and their Resilience to Socioecological Change, *Human Ecology* **39**(6)**:**769-781.

Synthèse partie III

La première partie de cette thèse s'est intéressée à la représentation et à la valeur attribuée aux services écosystémiques par les éleveurs (chapitre 3). La seconde partie a ensuite été consacrée à l'étude des changements de service en réponse aux modifications des pratiques de gestion dans différents scénarios climatiques et socio-économiques (Chapitre 5 et 6). Cette dernière partie examine comment les éleveurs considèrent ces changements et comment ceux-ci peuvent influencer leurs choix de pratiques, et de cette façon, elle boucle l'étude du système socio-écologique en explorant cet effet de rétroaction du changement de services sur l'utilisation du sol au travers des processus de décision des agriculteurs (Question 3).

Bien que le bienfondé des études des services écosystémiques par une approche socio-écologique soit souvent mentionné, encourageant notamment l'étude de l'effet de rétroaction des changements de services sur les choix de gestion des écosystèmes, la mise en œuvre de ce type d'étude est à notre connaissance novatrice. Celle-ci repose sur un jeu de rôle, réalisé avec les éleveurs de Villar d'Arène en janvier 2012, qui consiste à confronter les participants aux changements de services issus des adaptations qu'ils ont proposées dans le précédent jeu de rôle (Partie II) et d'observer leurs réactions en termes de changements de gestion des prairies. L'analyse des résultats nous a permis de comprendre leurs réactions aux changements en décrivant leurs processus de décision.

L'effet des changements de services sur le comportement relève majoritairement d'un feedback direct n'affectant que la décision elle-même sans modifier les connaissances et les valeurs. Il s'agit de décisions prises en réponse à des changements auxquels les éleveurs ont plus ou moins l'habitude de faire face, comme la diminution de la quantité de foin en cas de sécheresse. La conversion de la fauche au pâturage, constatée pendant le jeu, s'inscrit dans cette logique de feedback direct. Lorsque les composantes cognitives de la prise de décision (valeurs, connaissances) sont affectées par les changements de service ou des facteurs extérieurs, la décision relève d'un feedback indirect. Rarement observé, ce type de feedback peut être illustré par l'exemple de prise de décision d'arrêter la fertilisation grâce à un effet d'apprentissage de l'inefficacité de la fertilisation en cas de sécheresse.

Enfin, il est naïf de croire que les services écosystémiques influencent entièrement le comportement des agriculteurs. Celui-ci ne correspond pas toujours à leurs attitudes, par exemple en raison de contraintes liées à l'exploitation. Quoi qu'il en soit, ce cas d'étude montre que l'intérêt porté à un service écosystémique et/ou des connaissances à son égard est une condition nécessaire mais pas suffisante à sa prise en compte dans le choix de gestion des agriculteurs. Si certains services (ex. qualité et quantité de fourrage) contribuent aux choix de gestion, d'autres facteurs tels que la topographie, le contexte économique et social de l’exploitation, la possibilité de subventions peuvent amener l’agriculteur à prendre une décision en défaveur du service considéré. Ces résultats illustrent également le fait, que malgré la fourniture potentielle d’un service, l’agriculteur n’en bénéficiera pas forcément.

Partie IV

Chapitre 8

Discussion générale

1 Synthèse

Le développement et les applications du concept de services écosystémiques sont en plein essor depuis qu'il a été largement diffusé et promu par l'évaluation des écosystèmes pour le millénaire en 2005. Depuis lors, il a été intégré par différentes disciplines scientifiques, mais un domaine de recherche propre se focalisant sur l'étude des services écosystémiques est aussi en train d'émerger. Celui-ci cherche à comprendre et décrire, d'une part, comment et quels services écosystémiques sont fournis par différents écosystèmes (offre) et, d'autre part, quelles sont les attentes sociétales pour ces services (demande). Dans cette étude j'ai mis en place une approche socio-écologique pour l'étude des services écosystémiques sous ces deux aspects de fourniture et de demande, en l'appliquant à un socio-écosystème de prairies subalpines d'adret dans les Hautes-Alpes (France).

Dans le Chapitre 2, nous avons commencé par discuter l'appropriation et l'utilisation du concept de services écosystémiques au sein de la communauté scientifiques. L'augmentation du nombre d'études sur les services écosystémiques pourrait laisser penser que celui-ci est bien défini et qu'une approche standardisée est partagée entre chercheurs pour la réalisation d'études comparatives. Une analyse des principales publications sur le sujet a montré cependant que son interprétation diffère parmi les chercheurs, notamment selon leur discipline d'origine, ce qui peut conduire à des évaluations différentes en termes de qualité, de quantité ou de localisation des services et constituer un frein aux analyses comparatives entre étude. Si ce concept est encore flou au sein du domaine scientifique qu'en est-t-il pour les bénéficiaires, gestionnaires et décideurs politiques ? Le chapitre 3 de cette thèse a exploré, auprès des agriculteurs, des autres bénéficiaires locaux et des experts régionaux, la connaissance du concept de services écosystémiques dans le cas des prairies de montagne. Si le concept était encore peu connu au moment des enquêtes (2009-2010), celui-ci s'est avéré être un moyen efficace pour faire parler les gens des écosystèmes qui les entourent et de ce qu'ils en retirent. Une analyse comparative entre trois pays a montré que le classement par ordre d'importance des services par les enquêtés est dépendante du contexte et de la catégorie d'acteurs dont ils font partie. Toutefois, en zone de montagne, un groupe de services considéré important par les différents *stakeholders* des trois zones d'études nous a permis de mettre en évidence les services écosystémiques à étudier en priorité (la fertilité du sol, la production de fourrage, la conservation de la diversité botanique, l'esthétique, la qualité de l'eau et la séquestration du carbone). Ce premier volet a fourni les réponses à ma première question de recherche : « Quels sont les services écosystémiques perçus, utilisés et/ou appréciés par les *stakeholders* ? ».

Dans le chapitre 4, j'ai répondu à ma deuxième question de recherche (« Quel est le potentiel de fourniture des services par les prairies actuelles étant donné les dynamiques écologiques ? ») en modélisant la fourniture potentielle de ces services à l'aide d'un modèle spatialement explicite basé sur les traits fonctionnels. Cette approche a révélé les mécanismes écologiques qui déterminent les différents niveaux de fourniture de services entre types de prairies. Si l'adret de Villar d'arène fournit de multiples services, certains services sont néanmoins favorisés au détriment d'autres services en fonction de la gestion des prairies. Ceci peut conduire les *stakeholders* à orienter leurs décisions de gestion en fonction des services qu'ils considèrent les plus importants. Ces arbitrages ont fait l'objet des derniers chapitres qui étudient, d'une part, les capacités d'adaptation des agriculteurs (chapitre 5) et, d'autre part, la réponse de la fourniture de services écosystémiques (chapitre 6) en réponse à des changements climatiques et socio-économiques. Dans ce but, nous avons élaboré, conjointement avec les experts régionaux et les agriculteurs du site, des scénarios futurs d'utilisation du sol à un horizon temporel de 20 ans induits par des contextes climatiques et socio-économiques (chapitre 5). Les changements de gestion associés à ces quatre scénarios se sont avérés limités. Il s'agit principalement de l'augmentation des parcelles fertilisés et de la conversion de fauche en pâture des prés les moins mécanisables. La fourniture de services écosystémiques associée à la conversion fauche-pâture baisse de manière générale, en particulier les services culturels en raison de la baisse de la diversité floristique. Toutefois, ces changements n'affectent pas les liens entre services (*trade-off* et synergies) car ceux-ci dérivent de propriétés des écosystèmes et de traits fonctionnels dont les relations négatives et positives répondent de la même manière au sein des différents scénarios. Cette étude à donc répondu à ma troisième question de recherche : « Comment la gestion des prairies affecte la fourniture de services écosystémiques ? ».

Finalement, dans le chapitre 7, nous nous sommes penchés sur la manière dont les services écosystémiques sont pris en compte dans les processus de décisions de gestion d'utilisation du sol (question 4), sous l'angle de l'effet de rétroaction des changements de services sur l'utilisation du sol, selon les scénarios considérés (question 5 : « Comment le contexte global influence le système socio-écologique ?»). Cette étude montre que certains services contribuent à la prise de décisions des agriculteurs lorsque les connaissances et les valeurs attribuées à ces services par les agriculteurs sont révélées, mais ce sont surtout d'autres facteurs tels que la topographie, les valeurs sociales ou les caractéristiques de l'exploitation qui déterminent le comportement réellement adopté par l'agriculteur. Néanmoins, selon le contexte et les changements de services écosystémiques, les services pris en compte et leur contribution à la prise de décisions peut varier.

2 Principaux résultats

2.1 Sous-système écologique

Nous avons basé notre approche sur l'utilisation de traits fonctionnels végétaux et microbiens, comme cela est préconisé par d'autres études (de Bello et al., 2010; Diaz et al., 2007; Quétier, 2006), afin d'identifier quel est le potentiel de fourniture de services par les écosystèmes actuels du fait de leur diversité et de leur distribution dans le paysage (question 2) et de comprendre comment ces services réagissent aux changements d'utilisation du sol (question 3). Cette approche nous a permis d'utiliser des indicateurs précis (biotiques et abiotiques) pour distinguer et relier les différentes composantes écologiques de la cascade conceptuelle des services (Figure 1) : structures et processus, fonctions, et services écosystémiques (Chapitre 2). De plus, nous avons aussi pu projeter l'effet des changements climatiques ou d'utilisation des terres sur ces différentes composantes grâce à la concordance entre les traits de réponses aux facteurs environnementaux et les traits d'effets sur l'écosystème (Lavorel and Garnier, 2002). En modifiant les principaux paramètres abiotiques (fertilité du sol) et biotique (traits), cette approche mécaniste a permis de modéliser quantitativement les effets de changements climatiques et de gestion des prairies. Par rapport à une autre étude basée sur les composantes de la cascade conceptuelle des services mais utilisant des indicateurs liés à l'occupation du sol, la structure du sol et la végétation à une échelle locale (van Oudenhoven et al., 2012), nous pensons que l'utilisation d'indicateurs liés à la composition fonctionnelle des prairies apporte plusieurs avantages.

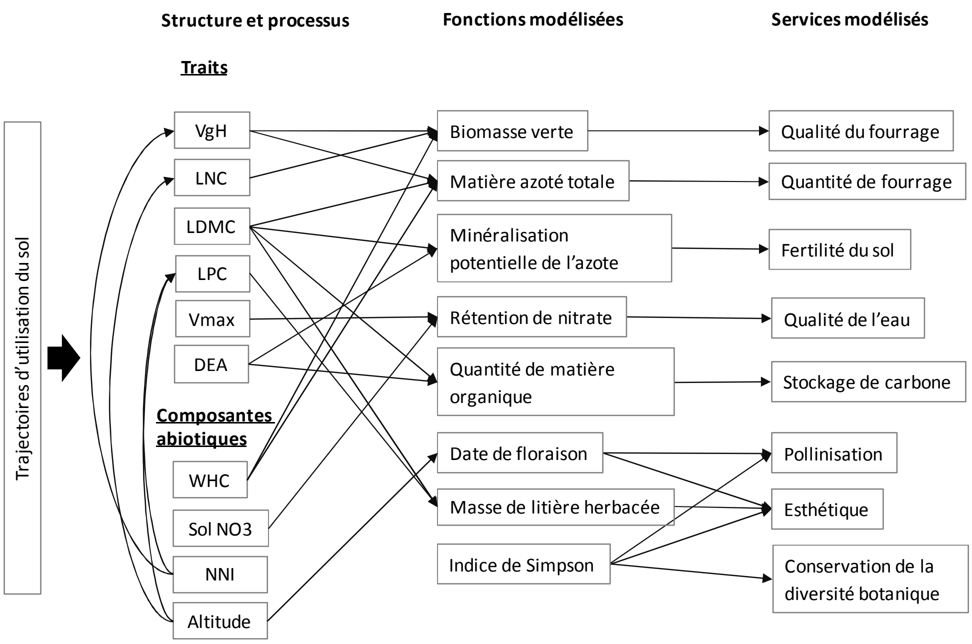


Figure 1 : Positionnement des données collectées et modélisées selon la cascade de Haines-Young et Potschin (2010) afin d'étudier les services des écosystèmes sur la commune de Villar d'Arène. VgH, Hateur de végétation ; LNC, contenu en azote des feuilles ; LDMC, contenu en matière sèche des feuilles ; LPC, contenu en phosphore des feuilles; Vmax, constante de l'activité de l'enzyme nitrifiante; DEA, Activité de l'enzyme denitrifiante ; WHC, Capacité de rétention en eau ; Sol NO3, teneur en azote du sol ; NNI, indice de nutrification azoté.

Premièrement, l'utilisation d'indicateurs à la fois cohérents par rapport à l'étude des services écosystémiques et flexible pour être appliqué à différents types d'écosystème ou dans différentes régions est importante (Niemeijer and de Groot, 2008). La diversité fonctionnelle et les traits semblent répondre à cet objectif par rapport à une approche basée sur la diversité spécifique car elle permet une comparaison entre sites de différentes régions et différents types d'écosystèmes où une végétation différente comporte pour autant les mêmes traits (Lavorel et al., 1997). Ces comparaisons pourraient être facilement mises en œuvre vu la disponibilité d'études et de données portant sur les traits fonctionnels mesurés par des protocoles standardisés (Cornelissen et al., 2003) dans de nombreux sites (Kattge et al., 2011). Si les modèles basés sur les traits requièrent de nombreuses données collectées à l'échelle locale, les bases de données traits existantes (Kattge et al., 2011; Kleyer et al., 2008) devraient permettre d'utiliser ce type de modèle à l'échelle régionale en couplant ces bases de données de traits avec des bases de données de végétation régionales.

Deuxièmement, les décideurs politiques et gestionnaires ont besoin d'informations claires sur la base desquelles ils peuvent identifier, prioriser et exécuter leurs interventions (Chapitre 2). Afin d'être utile en dehors de la recherche pour de multiples utilisateurs finaux les indicateurs doivent être clairs, compréhensibles et facilement mesurables (Niemeijer and de Groot, 2008).

Notre étude de la perception de la biodiversité avec les *stakeholders* (Chapitre 3) a suggéré que les traits, principalement les traits facilement mesurables (*soft traits*) peuvent avoir plus de sens pour eux que les espèces dont l'identification peut s'avérer difficile. Ces traits peuvent être facilement mesurés par des méthodes rapides et ne nécessitant pas de fortes connaissances botaniques, par les gestionnaires (Gaucherand and Lavorel, 2007), ou les agriculteurs (Duru et al., 2011; Fleury, 1997). De plus, l'utilisation de la diversité fonctionnelle par rapport à la diversité spécifique permet d'établir un pont entre l'approche scientifique et la manière dont les agriculteurs évaluent leurs végétations (Duru et al., 2011).

Troisièmement, les études prenant en compte uniquement l'utilisation ou l'occupation du sol pour évaluer les services écosystémiques se sont avérées peu précises (Eigenbrod et al., 2010). Les traits fonctionnels reflétant les effets fins de la gestion sur des variables explicites comme la fertilité, mais aussi de variables abiotiques telles que la topographie s'avèrent plus précis (Chapitre 4). En effet, cela permet de prendre en compte les effets au sein d'une catégorie d'occupation ou d'utilisation du sol tels que les prairies de fauche ou les alpages. Dans cette thèse, nous avons contribué en partie à combler le manque d'étude sur la fourniture de services écosystémiques au sein d'une catégorie d'utilisation du sol selon des changements d'intensité de gestion (Bennett et al., 2009; Rounsevell et al., 2012) en prenant en compte les trajectoires d'utilisation du sol (Chapitres 4 et 6). Néanmoins, dans cette thèse nous avons étudié cet effet uniquement de manière discrète (fauche vs pâture, fertilisé vs non fertilisé), sans pouvoir prendre en compte des variables plus fines telles que le chargement animal ou la date d'utilisation. Cependant, les traits semblent être un outil adéquat pour analyser l'effet d'un gradient d'intensité de fertilisation ou de pâturage par le taux de chargement (Bluthgen et al., in press; Duru et al., 2005; Duru et al., 2012b).

Quatrièmement, notre étude prospective sur l'effet du climat et/ou de la gestion des prairies a permis par l'utilisation des traits de prendre en compte des changements de services écosystémiques plus fins que des changements de composition floristique l'auraient permis (Chapitre 6), notamment grâce à leur variabilité inter- et intra-spécifique (Albert et al., 2010; de Bello et al., 2011). Elle a aussi permit de comprendre les mécanismes qui sous-tendent la fourniture de chacun des services, et d'identifier les causes de changements simultanés de multiples services en réponse au changement climatiques et de gestion des prairies. En effet, dans les prairies subalpines le principal facteur limitant est l'azote (Robson et al., 2010), hors la disponibilité des nutriments du sol affectée par le climat et/ou le changement de gestion (fauche vs pâture, fertilisation) va modifier la composition des communautés favorisant la dominance d'espèces conservatrices (ex. moins de nutriments disponible lors de sécheresse ou de la conversion de la fauche vers la pâture) ou d'espèces exploitatrices (plus de disponibilité en nutriments) (Quétier et al., 2007b). La façon dont se comportent les traits entre les stratégies exploitatives et conservatives (c.-à-d. compromis ou synergies), se retrouve au niveau des services écosystémiques par l'intermédiaire des propriétés sur lesquels les traits agissent (Lavorel and Grigulis, 2012). Autrement dit, les compromis et synergies de services sont contrôlés par ces mêmes relations au niveau des traits.

2.2 Sous-système social

Pour étudier les bénéfices que les acteurs locaux obtiennent des écosystèmes (question 1) et la façon dont ceux-ci influencent les décisions des acteurs (question 4), nous avons fait l'hypothèse, tout comme Roling and Wagemakers (1998), que les perceptions que les gens ont de leur environnement déterminent leurs actions sur et dans cet environnement, ces perceptions pouvant être modifiées par une meilleure compréhension de cet environnement et par les interactions avec les autres parties prenantes. Cette hypothèse est confortée par (Meyfroidt, 2012) qui suggère que les modélisations de décisions (ex. modélisation multi-agents (Boone et al., 2011)) devraient prendre en compte la variabilité de la rationalité dans les prises de décision due aux perceptions, connaissances et valeurs des individus qui peuvent changer selon le contexte et dans le temps.

C'est pourquoi, nous avons décidé d'ajouter la dimension des perceptions et des connaissance des services écosystémiques à la dimension des valeurs déjà présente dans les cas d'étude sur les services (Brondizio et al., 2010). A cette fin, nous avons adopté une approche participative auprès des agriculteurs et des experts régionaux celle-ci étant appropriée pour collecter des informations sur les perceptions des *stakeholders* et particulièrement celles qui sont difficilement quantifiables, mais aussi pour mettre en évidence les différences et similitudes entre *stakeholders* (Mendoza and Prabhu, 2005). L'implication des *stakeholders* dans les processus de recherche a été reconnu essentiel afin de définir, identifier et décrire les services dans leurs propres termes pour comprendre le système en question (Diaz et al., 2011).

Le chapitre 7 a bien montré l'intérêt d'inclure les perceptions et les connaissances des *stakeholders* dans l'analyse du processus de décision des agriculteurs. Ces deux composantes ainsi que les valeurs influencent effectivement leurs prises de décisions de gestion des terres, sans pour autant être déterminantes (conditions nécessaire). Nous ne pouvons supposer que l'ensemble de la population soit informé, conscient et en mesure de comprendre tous les liens complexes entre l'écosystème, les paramètres biophysiques et les activités humaines, alors que ceux-ci constituent encore un défi pour la recherche (Costanza, 2008). Toutefois notre étude a montré que les connaissances des agriculteurs, et experts régionaux peuvent s'avérer riches et diverses notamment pour les liens entre pratiques et services écosystémiques liés aux plantes. Les connaissances et valeurs associées aux propriétés et services liés au sol et à sa biodiversité sont par contre moins bien connus, comme cela a souvent été vérifié dans d'autres études (Barrios, 2007), excepté au travers des effets de fertilisation (chapitre 1). Cette étude a aussi mis en évidence l'existence de variations de perceptions, de connaissances et de valeurs entre *stakeholders*, types de *stakeholders* et entre régions (chapitre 1). De plus, si nous avions déjà identifié que les individus n'avaient pas une préférence fixe par rapport à un groupe de services selon qu'ils se placent par rapport à leur vision personnelle ou professionnelle (chapitre 1), nous avons aussi montré que les valeurs, connaissances et perceptions pouvaient varier selon le contexte, le partage d'information avec les autres agriculteurs, les experts régionaux et les chercheurs (chapitre 7). Ceci souligne l'importance de combiner des entretiens individuels, avec des méthodes collectives (entretiens collectifs ou jeux de rôles), mais aussi l'intérêt d'adopter une démarche itérative dans laquelle les activités de restitution, de consultation et de dialogue alternent et s'enrichissent mutuellement. Cet apprentissage mutuel entre chercheurs et *stakeholders* a offert la possibilité d'approfondir au cours du temps les discussions sur certains aspects des services écosystémiques, notamment sur les relations entre eux (Chapitre 7).

2.3 Liens entre sous-systèmes et effets de rétroaction

Notre approche socio-écologique repose sur l'étude des liens entre les deux sous-systèmes et combine à cet effet des données qualitatives sur les discours de *stakeholders* et des données quantitatives sur les processus et fonctions des écosystèmes. Ces liens sont assurés par les pratiques agricoles et les services écosystémiques et nous avons cherché à voir comment ces liens, et les deux sous-systèmes, co-évoluent en cas de changements (comme Tengö and Belfrage, 2004). Mon dispositif expérimental a consisté en un jeu de rôle associant des données d'utilisation du sol et de services traduites sur un plateau de jeu, à une analyse des discours des agriculteurs sur les motifs de leurs décisions durant le jeu et les débriefings des parties.

Les résultats ont montré que les changements de services affectent les décisions des agriculteurs au travers de deux types d'effets de rétroaction (Meyfroidt, in revision). Le premier type, majoritairement observé est direct car il n'affecte que la décision elle-même sans modifier les connaissances et les valeurs. Il s'agit de décisions prises en réponse à des changements auxquels les éleveurs ont plus ou moins l'habitude de faire face, comme la diminution de la quantité de foin en cas de sécheresse. La conversion de la fauche au pâturage, constatée pendant le jeu, s'inscrit dans cette logique de rétroaction directe. Le deuxième type relève d'une rétroaction indirecte car les composantes cognitives de la prise de décision (valeurs, connaissances) sont affectées par les changements de service ou des facteurs extérieurs. Rarement observé, ce type de rétroaction peut être illustré par la prise de décision d'arrêter la fertilisation grâce à un effet d'apprentissage de l'inefficacité de la fertilisation en cas de sécheresse (Chapitre 7).

Cette prépondérance d'effet de rétroaction direct est probablement liée aux caractéristiques du système socio-écologique étudié. En effet, le système étudié fournit de nombreux services écosystémiques entre lesquels les compromis existent principalement entre les services liés au sol moins visibles (fertilité du sol, stockage de carbone, qualité de l'eau) et les services liés à la végétation plus visibles (quantité du fourrage, esthétique, conservation de la diversité botanique), hormis dans les pâturages sur les anciennes prairies de fauches envahis par la Fétuque paniculée au sein des quels la qualité du fourrage et la diversité floristique sont fortement diminuées (Chapitre 6).

Or, les services du sol étant considérés comme peu importants par les agriculteurs, à l'exception de la fertilité du sol souvent confondue avec la fertilisation (Chapitre 2), ces compromis n'amènent pas les agriculteurs à devoir arbitrer leurs décisions en faveur d'un service. Cette arbitrage intervient uniquement si ces services de régulation tels que le stockage de carbone ou la qualité de l'eau sont privilégiés par les décideurs politiques au niveau national ou européen et découlent sur des paiements pour services écosystémiques (Chapitre 7) conduisant à un feedback indirect. Actuellement, ce type de paiement n'existe pas et ce compromis de services n'entraine donc pas de conflit entre catégories de *stakeholders*.

Cependant, le système socio-écologique peut évoluer en fonction du contexte climatique et socio-économique. Selon nos résultats, les principaux changements sur le système écologique et la fourniture de services sont dus aux effets directs du climat (scénario drastique) conduisant à un renversement de la fourniture en services favorisant les services liés au sol au détriment des autres services (Chapitre 6). Notre étude montre que les agriculteurs ont des difficultés à réagir à ces changements en raison des contraintes imposées par le système montagnard (altitude, topographie, durée de végétation courte). Cette simulation nous a fait prendre conscience ainsi qu'aux agriculteurs que, contrairement à d'autres systèmes (Nettier et al., 2010; Tengö and Belfrage, 2004), leurs systèmes d'élevage actuels comportent peu de marge de manœuvre pour faire face à des changements importants. Les faibles effets de la fertilisation en cas de sécheresse réduisent leur marge de manœuvre pour contrer la perte de production agricole (faible augmentation de quantité, mais diminution de qualité). De plus, les adaptations visant à convertir les prairies fauchées en pâtures, surtout si elles sont maintenue à long terme, pourraient renforcer cette dégradation en diminuant fortement la qualité du fourrage et la diversité floristique (Chapitre 6). L'étroite marge de manœuvre et les décisions prises se traduisent donc par un effet de rétroaction négatif sur la fourniture de services, ce qui réduit l'offre par rapport à la demande de services. Nos résultats montrent donc une faible co-évolution entre les pratiques agricoles et les services écosystémiques. D'un côté, les changements de pratiques ont réduit l'offre de services souhaités et d'un autre côté, les agriculteurs n'ont pas pu adapter leurs pratiques pour faire coïncider offre et demande de services. Dans cette configuration, la résilience du système ne sera probablement pas assurée sans l'influence de facteurs extérieurs (ex. aides agricoles).

2.4 Apports et limites d'une approche socio-écologique des services

Les études de quantification et modélisation spatiale des services écosystémiques ne font généralement pas la distinction entre l'approvisionnement potentiel (les fonctions) et la demande en services écosystémiques (Chapitre 2). L'approche socio-écologique adoptée pour le cas d'étude de Villar d'arène a permis d'étudier, d'un côté, l'approvisionnement (Chapitres 4 et 6) via le système écologique et, d'autre côté, la demande (Chapitres 3 et 7) à travers le système social, mais surtout de joindre ces deux composantes notamment par l'analyse de l'effet de rétroaction du changement de services sur la prise de décision des acteurs, le tout dans un contexte dynamique (Chapitre 5). Cette étude sur les effets de rétroaction n'aurait pas été possible si la fourniture et la demande de services écosystémiques n'avaient pas été étudiées toutes deux en amont.

Cette analyse des prairies comme un système socio-écologique intégré et l'étude des effets de rétroaction rendent possible le passage d'une analyse centrée sur les réactions aux changements observés à une analyse proactive qui intègre la durabilité à long terme de l'effet des pratiques agricoles sur la fourniture de services écosystémiques. De plus, l'implication des *stakeholders* dans la description du système et ses points sensibles, notamment par l'intermédiaire de développement de scénarios prospectifs a révélé les principales évolutions des services ecosystémiques. L'utilisation de modèles décrivant l'évolution du système écologique sous ces différents scénarios permettrait d'identifier les composantes de la résilience du système et comment celles-ci peuvent être perdues ou renforcées (Walker et al., 2002). Par la compréhension de ces dynamiques, les politiques peuvent renforcer les capacités adaptatives des exploitations agricoles vers une fourniture de services écoystémiques souhaitées, notamment dans les situations de changement inattendu.

Ces connaissances de la dynamique interne du système socio-écologique doivent être élargies à la compréhension de l'influence de facteurs extérieurs au système. Ce besoin de prendre en compte les effets de l'influence de facteurs émanant de l'extérieur du système socio-écologique (socio-économique, climat, …) sur la demande et la fourniture de services a été pointé par notre étude prospective. Nous n'avons cependant pris en compte ces éléments extérieurs que par leur rôle sur le fonctionnement du système au sein de la démarche prospective, sans s'occuper de l'effet du système lui-même sur ces éléments, ce qui confère à notre étude un caractère relativement fermé. Pourtant, les systèmes socio-écologiques devraient être étudiés et modélisé comme des systèmes ouverts intégrant les flux globaux de services écosystémiques, de personnes et de capitaux vers l'intérieur et l'extérieur du système local (Lambin and Meyfroidt, 2011). Par exemple, l'influence du changement d'attentes des consommateurs peut affecter indirectement les systèmes agricoles de Villar d'Arène. Si la demande en Beaufort venait à diminuer, les éleveurs savoyards ne pourraient probablement plus faire élever leurs génisses par les agriculteurs de Villar d' Arène. De plus, dans cette étude, j'ai exploré uniquement les effets de rétroactions à l'échelle de la prise de décision des acteurs. Cependant, par les mécanismes de boucles de rétroactions intervenant entre et à de multiples échelles, les changements d'utilisation du sol et donc de services à une échelle locale, peuvent être répercuté à l'échelle globale et entrainer des réponses institutionnelles (Carpenter et al., 2009; Lambin and Meyfroidt, 2011; Rounsevell et al., 2012).

Pour analyser ces différents aspects d'interaction entre l'Homme et son environnement, les recherches sont nécessairement de nature interdisciplinaire (Young et al., 2006), voir même transdisciplinaire car l'intérêt de l'implication des *stakeholders* est de plus en plus démontré notamment dans l'objectif d'un développement durable (Duru et al., 2012a; Rounsevell et al., 2012). Ce partage de connaissances du système entre chercheurs de différentes disciplines et de *stakeholders* constitue un des avantages de l'approche socio-écologique (Huitric et al., 2009).

Les approches participatives peuvent impliquer les parties prenantes à différents degrés allant de la simple fourniture d'information sur les résultats de recherche à la co-construction de modèle de recherche (Barreteau et al., 2010a; Volkery et al., 2008). Dans cette étude, la participation des *stakeholders* a été au-delà de la simple information sur les résultats de recherche (Annexe 1) ou de consultation pour la collecte de données. Au travers d'un processus itératif, l'échange d'information a été à double sens entre chercheurs et *stakeholders* par l'intermédiaire d'une combinaison d'outils participatifs tels que des entretiens individuels ou collectifs, de cartographie mentales ou de jeux de rôles (Barreteau et al., 2010a). Cette approche a permis d'améliorer notre connaissance du système socio-écologique en confrontant notre vision et la vision des *stakeholders*. De plus, la combinaison de notre représentation quantitative et de leurs connaissances qualitatives a souligné l'intérêt de ce type d'approche et de sa rigueur scientifique, bien que des critiques existent sur la dimension qualitative de ces approches et le manque apparent de rigueur systématique pour interpréter les données fournies par les *stakeholders* (Mendoza and Prabhu, 2005).

Il n'en reste pas moins que l'implication de *stakeholders* dans les projets de recherches amène à relever certains défis. Afin de maintenir leurs implications et leurs motivations tout au long du projet, il faut veiller à formuler des questions pertinentes par rapport à leurs propres intérêts. Ceci est d'autant plus important que le temps qu'ils consacrent au projet peut être conséquent

(Walz et al., 2007). A cet égard, il est donc aussi judicieux d'adopter des méthodes flexibles et qui requièrent le moins de temps possible. Par ailleurs, il s'avère utile de formuler en des termes compréhensibles les concepts scientifiques utilisés, comme la biodiversité ou les services écosystémiques. A cette fin, nous avons fait un effort particulier pour traduire la complexité des multiples paramètres socio-écologiques pris en compte dans notre approche en utilisant des outils variés et le plus illustratifs possibles, notamment par l'intermédiaire de jeux dont la pertinence pour la conversion de concepts scientifiques sous des formes compréhensibles par les *stakeholders* a déjà été soulignée dans d'autres études (Barreteau et al., 2001; Martin et al., 2011).

Mais les difficultés des approches transdiciplinaires résident dans l'intégration des différentes disciplines de recherches. La transdisciplinarité de l'approche ne doit pas diminuer la rigueur des différentes disciplines mobilisées (Diaz et al., 2011). Au contraire, elle doit se construire sur les atouts des disciplines impliquées au travers de la combinaison des connaissances diversifiée de chacune d'elle. N'existant pas de procédures standards, les approches interdisciplinaires doivent se construire sur une pluralité de centres d'intérêts et de méthodes (Bammer, 2012). Des méthodes et outils adéquats doivent être utilisés pour étudier chaque composante du système socio-écologique. C'est pourquoi nous avons adopté et couplé des méthodes provenant de l'écologie (modélisation statistique basée sur les traits fonctionnels), des sciences sociales (interview, enquête, jeu de rôle), de la géographie (utilisation de systèmes d'information géographique, analyse du paysage) ou de l'agronomie (analyse des systèmes d'exploitation). Pour répondre aux exigences de l'interdisciplinarité, nous avons dû adopter des techniques d'extrapolation ou d'agrégation des données pour résoudre certains problèmes liés à la mise en commun des échelles spatiales des données inhérentes aux différentes disciplines, allant du point d'échantillonnage à la région (Mottet et al., 2007). Cette tâche a été facilitée par la disponibilité de nombreuses données biotiques et abiotiques collectées depuis 2003 par un échantillonnage stratifié selon l'altitude, les types de prairies et les secteurs du versant étudiés, mis en place dans le cadre de la Zone Ateliers Alpes (site d'étude de recherche à long terme). Dans le même ordre d'idée, les pas de temps entre l'évolution écologique et l'évolutions des activités humaines ne sont pas du même ordre de grandeur, ce qui rend plus difficile la mise en œuvre des études prospectives, surtout si les scénarios se veulent cohérents par rapport au décisions politiques qui ont lieu à court-terme (Rounsevell et al., 2012).

Pour conclure cette section nous mentionnerons certaines critiques philosophiques et éthiques citées dans la littérature à propos de l'approche socio-écologique et de la vision anthropocentrée du concept de services écosystémiques. En effet, cette représentation continue de séparer les sociétés humaines et les écosystèmes reflétant la vision occidentale de la place de l'Homme dans l'écosystème planétaire (Barnaud et al., 2011). Hors selon Hansson et Wackernagel (1999), cette séparation de l'homme et la nature serait à l'origine des problèmes de dégradation de l'environnement. Personnellement, je pense que l'approche socio-écologique des services écosystémiques participe à un premier pas vers un rapprochement des systèmes sociaux et écologiques en un seul système.

3 Implications pour les politiques publiques et le conseil auprès des agriculteurs

Cette étude a montré que les écosystèmes prairiaux de haute montagne fournissent actuellement de multiples services écosystémiques dont bénéficient différents *stakeholders* (ex. agriculteurs, habitants de la commune ou de la région, touristes, la société). Actuellement, la fourniture de ceux-ci dépend de l'utilisation du sol et de son intensité (fauche, pâture, fertilisation). Mais ces écosystèmes sont aussi soumis à des transformations induites par les modifications de pratiques agricoles, elles-mêmes découlant des changements climatiques et/ou socio-économiques. Ces changements de pratique, et de services écosystémiques associés, restent toutefois limités à l'intérieur d'une même catégorie d'utilisation du sol (prairies) malgré les contrastes importants des scénarios climatiques et socio-économiques. Les contraintes physiques du site (climat, topographie, risques naturels) limitent en effet fortement la marge de manœuvre des agriculteurs pour le changement de pratique ni d'envisager autre chose que l'agriculture. Dans notre étude, le changement le plus important est l'arrêt de la fauche en faveur du pâturage sur l'ensemble de l'adret (scénario Drastic-Local), afin de limiter les coûts de production élevés inhérents au système de haute montagne. Cette conversion vers le pâturage avait déjà été observé dans une autre étude (Quétier, 2006) où un système de transhumance inverse avait été proposé. Dans un tel système, les éleveurs posséderaient de grands troupeaux pour dégager un revenu suffisant et iraient mener l'hiver les bêtes indigènes des alpes sur d'autres pâturages plus au Sud (Arbos, 1920). Ces changements pourraient même avoir lieu dans un futur proche, si les départs en retraite ne sont pas compensés par l'installation de nouveaux exploitants sur la commune. Les terrains les moins favorables à la mécanisation seraient alors probablement pâturés au détriment de la fauche. Or la fauche joue un rôle crucial dans le maintien d'un niveau élevé de biodiversité (Fischer and Wipf, 2002; Rudmann-Maurer et al., 2008). Cet effet se vérifie particulièrement, sur notre site, dans les prairies en dehors des zones de terrasses en créant un niveau de perturbation intermédiaire qui empêche la colonisation de touffes dense d'herbes indésirables telles que la Fétuque paniculée (*Festuca paniculata*) ou, dans une moindre mesures sur les terrasses, qui empêche la dominance du Brome érigé (*Bromus* *erectus)* (Quétier et al., 2007a). Ceci conduit à une diminution de la qualité du fourrage, de la diversité floristique et de la fertilité des sols (Chapitre 6) (Quétier et al., 2007a). Un pâturage très extensif ou un abandon des terres pourrait même mener en 50 ans à un reboisement par l'établissement de population de mélèzes (*Larix decidua*) (Albert et al., 2008) tel que cela a été observé pour d'autres espèces ligneuses dans les Pyrénées (*Fraxinus excelsior* (Mottet et al., 2007)), ou dans les Alpes autrichiennes (Tasser et al., 2007), suisses (Gellrich et al., 2007) et italiennes (Cocca et al., 2012). Au détriment des services fournis actuellement par les prairies, le reboisement pourrait conduire à la fourniture d'autres services écosystémiques tel que la réduction d'avalanche (Teich and Bebi, 2009) ou une renforcement de la fourniture de certains services tel que l'augmentation de la quantité de carbone stocké (Tappeiner et al., 2008b).

Dans ce contexte de faible potentialité de changement d'occupation du sol (ex. terre arable), la principale menace est donc l'arrêt de la fauche en faveur de la pâture sur les parcelles les moins mécanisables. Afin de restreindre ces changements, une aide incitative peut-être mise en place pour compenser les agriculteurs adoptant ou maintenant des pratiques en faveur de la fourniture des services écosystémiques souhaités ou encourager ceux qui autrement adopteraient des pratiques à l'encontre de la fourniture de ces services (Sommerville et al., 2009). En effet, nos résultats ont montré qu'en dehors d'incitations financières ce sont principalement les services marchands (ex. qualité et quantité du fourrages) ou intrants (fertilité du sol) (selon la classification de Zhang et al., 2007) qui influencent la prise de décisions des agriculteurs, bien que d'autres facteurs (ex. topographie, distance) ou des *trade-offs* entre services peuvent les amener à prendre une décision qui va à l'encontre de la fourniture de ces services. Cette différence entre bénéfices privés et sociaux menant à des choix de gestion en faveur de peu de services est classique (Jack et al., 2008). Les agriculteurs ne percevant pas les bénéfices de ces services non marchands, des payements pour services écosystémiques peuvent être proposés par les bénéficiaires (société par l'intermédiaire de fond européen, national ou local) aux agriculteurs qui assument les coûts liés à la production de ces services (Sommerville et al., 2009). Le montant du dispositif de payement doit prendre en compte non seulement les coûts de la pratique agricole favorable aux services écosystémiques (ex. coûts des charges additionnelles de la fauche sur des terrains en pente par rapport à la pâture) à verser à l'agriculteur, mais aussi les coûts liés à sa mise en place (c.-à-d. l'évaluation de la fourniture de ces services, de l'effet de la pratique et des contrôles du respect de cette pratique) (Jack et al., 2008). La somme de ces coûts indique le montant que les décideurs politiques sont prêts à engager pour maintenir le service écosystémique considéré, ce qui revient à lui attribuer une valeur monétaire. Ce type de payement suppose de bien prendre en compte les compromis entre services et de garantir une coordination entre les différentes politiques publiques et les dispositifs de payement pour services écosystémiques définis à différentes échelles de décision politique (local à européen) (Kemkes et al., 2010) afin de ne pas amener les éleveurs à évaluer les coûts d'opportunité induits par chaque dispositif (Vatn, 2010). Ceci peut se traduire par la contractualisation de l'aide qui serait la plus avantageuse financièrement au détriment des services importants pour les bénéficiaires locaux, ou par l'adoption d'aides qui vont à l'encontre des sentiments moraux envers la conservation de certains services par l'agriculteur lui-même.

Par ailleurs, les exigences liées aux subventions représentent aussi un élément important. En effet, le passage de la fauche à la pâture sur les parcelles les moins mécanisables a principalement eu lieu dans l'alternative socio-économique globale car elle était peu contraignante en terme de pratiques et de résultats. Actuellement, les payements dans le cadre des subventions européennes agri-environnementales s'appliquent généralement aux actions (pratiques agricoles) favorisant la fourniture de services plutôt qu'aux résultats (Gibbons et al., 2011). La plus grande flexibilité de gestion des prairies est un des principaux arguments en faveur des aides visant à des résultats (Schwarz et al., 2008). En effet, la flexibilité des pratiques est particulièrement importante dans les systèmes agricoles très extensifs utilisant peu d'intrants pour faire faire face à des événements imprévus (ex. climat, campagnols). Ces aides sont plus difficiles à mettre en œuvre car il faut trouver des indicateurs de résultats fiables et mesurables sur le terrain (par exemple la présence de certaines espèces (Plantureux et al., 2010; Wittig et al., 2006), mais également par l'existence d'un décalage temporel entre la mise en place des pratiques et l'obtention des résultats souhaités. De plus, ce type de subventions nécessite une connaissance des agriculteurs de l'effet de leurs pratiques sur les services à obtenir. Nos résultats ont montré que leurs connaissances basées sur leur expérience de terrain étaient relativement bonnes pour tous les services « visibles » (ex. tel que la quantité de fourrage lors de la récole, ou la qualité du fourrage par l'appétence qu'en a le bétail ou encore l'esthétique par le nombre de fleurs), alors que les services moins visibles tels que les services liés au sol (stockage de carbone, fertilité naturelle du sol, qualité de l'eau) sont moins bien connus. L'effet de pratiques dans des contextes climatiques non encore observés fait aussi partie des connaissances à partager. Par exemple, certains agriculteurs avaient fertilisé dans le cas des sécheresses drastiques pensant augmenter la production de fourrage, alors que notre modélisation n'a pas indiqué d'effet de cette pratique dans ce scénario. Nos résultats ont montré que la connaissance et les valeurs associées aux services influencent la prise de décision des agriculteurs. Par conséquent, l'information et la prise de conscience autour des services écosystémiques devrait faire partie intégrante des subventions liés aux résultats. Cette information peut-être faite à l'aide d'outils cognitifs qui sont des moyens d'apprentissage qui facilitent et stimulent le processus cognitif en utilisant un support technologique et jouent le rôle de médiateur entre agriculteurs, techniciens et/ou chercheurs (Duru and Martin-Clouaire, 2011). De plus, la contractualisation de ce genre d'aide peut être freinée par les attitudes des agriculteurs envers le risque de ne pas atteindre l'objectif (Nettier et al., 2011), notamment lorsque un effort important doit être réalisé pour atteindre les objectifs sans certitude de les atteindre (Gibbons et al., 2011). L'utilisation d'un jeu tel que nous l'avons mis en place peut être utilisé aussi comme un support d'aide à la décision, en permettant aux agriculteurs de tester l'effet de leurs pratiques face à différentes situations et de discuter les résultats avec les techniciens agricoles pour les ajuster aux mieux et enlever une part d'incertitude en anticipant les résultats. Que se soient dans notre étude ou d'autres études (Duru et al., 2012a; Etienne and (coord), 2010), les agriculteurs se sont montrés généralement intéressés par ce genre d'approche d'apprentissage active de construction de connaissance par rapport à un apprentissage passif lors de présentation par une personne compétente.

4 Perspectives et conclusion

Plusieurs pistes de recherche pour l'avenir émergent de cette thèse.

Dans mon travail, la fourniture potentielle de services écosystémiques a été cartographiée par une démarche basée sur les interactions verticales entre les valeurs des différents éléments de la diversité, de l'utilisation du sol ou de la topographie sous-tendant la fourniture de services à l'échelle du pixel ou de la parcelle agricole. La prise en compte des interactions horizontales basées sur la diversité et la configuration paysagère ouvrirait, quant à elle, des perspectives intéressantes à notre travail. En effet, une approche verticale ignore les interactions importantes entre éléments du paysage (Fahrig et al., 2011; Tscharntke et al., 2005) mais aussi les différences de fourniture de services selon la position des éléments dans le paysage (ex. mosaïque de bosquets et pelouses pour l'habitat du Tetras-lyre (Schweiger et al., 2012) ou position des forêts sur la pente pour la diminution du risque d'avalanches (Teich and Bebi, 2009)). Une analyse de photographies aériennes ou images satellites combinée à l'utilisation de mesures de fragmentation serait une piste intéressante en prolongement de notre étude.

Cette perspective paysagère est également importante pour étudier les relations entre la localisation des bénéficiaires et le lieu de fourniture potentielle des services (Egoh et al., 2009; Hein et al., 2006). Les localisations de la fourniture et du bénéfice peuvent effectivement être identiques ou différentes. Un service peut être utilisé sur place (ex. quantité de foin, esthétique), à proximité de l'écosystème (ex. pollinisation), dans la direction du flux (ex. qualité de l'eau), indépendamment de la localisation (ex. séquestration du carbone), en fonction des déplacements des bénéficiaires (ex. activités récréatives, esthétique) (Costanza, 2008). Certaines études ont exploré ces aspects spatiaux entre localisation des bénéficiaires et lieu de fourniture. Par exemple, des données sur la présence d'infrastructures sont parfois utilisées pour mettre en relation services et bénéficiaires, comme par exemple les routes et points de vue à proximité des écosystèmes fournisseurs de services esthétiques (Reyers et al., 2009; Willemen et al., 2008). Une autre étude a montré les relations de partage transfrontalier de services entre un pays fournisseur et un pays utilisateur (Lopez-Hoffman et al., 2010).

Un autre axe de développement concerne les représentations spatiales de la fourniture de services et la localisation de la demande en services décrites par les acteurs eux-mêmes, qui sont rarement explorées (Alessa et al., 2008; Raymond et al., 2009). Cette question pourrait être illustrée sur notre site grâce à l'étude des échanges de services entre les régions de plaine et de montagne. Ces échanges pourraient concerner les services de production agricoles recherchés par les éleveurs dans le cas de transhumance (« classique » ou inverse), ou les services de régulation telle que la qualité de l'eau dont bénéficient les populations en aval. L'utilisation d'outils de visualisation semble particulièrement pertinente pour étudier les relations spatiales entre demande et fourniture de services écosystémiques. Par exemple, des approches participatives basées sur des techniques de cartographies cognitives ou mentales (McKenna et al., 2008) ou de visualisation du paysage en trois dimensions (Griffon et al., 2011) avec la possibilité d'identifier des espèces végétales structurantes (http://www.lvml.net/) permettrait de révéler quels bénéficiaires identifient quels services en quels endroits sur base de quels critères.

Notre étude sur la rétroaction du changement de services était principalement exploratoire. Celle-ci pourrait être approfondie par une approche combinant un jeu de rôle et un modèle multi-agents spatialement explicite ou une simulation multi-agents participative où les participants prennent le contrôle des agents (Guyot and Honiden, 2006). Ce type de modélisation adoptée par le groupe CommMod (Etienne and (coord), 2010) combine généralement une représentation de l'évolution de l'environnement et de ses ressources par automate cellulaire et un système multi-agent qui simule le comportement des *stakeholders* en fonction des conditions environnementales et socio-économiques selon des règles définies. Les systèmes multi-agent autorisent la simulation spatiale et temporelle du fonctionnement d'un système socio-écologique au niveau de l'exploitation agricole (Le et al., 2008), de l'effet des décisions sur les services écosystémiques (Boone et al., 2011), ainsi que de l'effet de rétroactions des services sur les décisions (Murray-Rust et al., 2011). Couplé à un jeu de rôle, ceux-ci donne l'opportunité de montrer en temps réel l'effet des choix de gestion sur la fourniture de services écosystémiques et de faire un jeu itératif sur l'effet de décision à plusieurs pas de temps (si les données écologiques sont disponibles à plusieurs pas de temps, ce qui n'était pas notre cas). Les modèles multi-agents offre la possibilité de combiner les résultats des modèles de décisions ascendants (*stakeholders* vers chercheurs) fournissant des informations sur les réponses et l'adaptation des agents aux conditions de changements environnementaux ou socio-économiques, avec des modèles écologique, économique ou politique descendants (chercheurs vers *stakeholders*) prenant en compte des facteurs de changements à l'échelle locale et supra-locale (Rounsevell et al., 2012). Ce type d'approche aurait donc l'avantage de prendre en compte pour plusieurs scénarios, les relations entre de nombreuses composantes du socio-écosystème, y compris les décisions d'acteurs autres que les agriculteurs (ex. touristes, gestionnaire du Parc National) tout en étudiant l'effet du changement de services à l'échelle individuelle de l'exploitant.

Pour conclure, les résultats de cette étude ont contribué aux fondements de la recherche empirique sur les services écosystémiques par la collecte et l'analyse de données qualitatives et quantitatives à l'échelle locale pour le cas d'étude de Villar d'Arène. Ces résultats ont montré que les services écosystémiques vont au-delà d'un concept à la mode inutilisable et peuvent s'avérer être un moyen d'améliorer les efforts envers la conservation de la nature. Par la mise en évidence des interactions homme-environnement et son rôle de support d'un véritable dialogue transdisciplinaire, ce concept semble prometteur pour mieux comprendre les interdépendances du système complexe dont l'homme fait partie. Toutefois, cela suppose l'utilisation d'une approche basée sur un pluralisme méthodologique dont cette étude a exploré la faisabilité.

5 Références

Albert, C. H., Thuiller, W., Lavorel, S., Davies, I. D., Garbolino, E., 2008, Land-use change and subalpine tree dynamics: colonization of Larix decidua in French subalpine grasslands, *Journal of Applied Ecology* **45**(2)**:**659-669.

Albert, C. H., Thuiller, W., Yoccoz, N. G., Soudant, A., Boucher, F., Saccone, P., Lavorel, S., 2010, Intraspecific functional variability: extent, structure and sources of variation, *Journal of Ecology* **98**(3)**:**604-613.

Alessa, L., Kliskey, A., Brown, G., 2008, Social-ecological hotspots mapping: A spatial approach for identifying coupled social-ecological space, *Landscape and Urban Planning* **85**(1)**:**27-39.

Arbos, P., 1920, La transhumance savoyarde en Provence, *Revue de géographie alpine* **8**(4)**:**665-666.

Bammer, G., 2012, trengthening Interdisciplinary Research: What it is, what it does, how it does it and how it is supported, Report for the Australian Council of Learned Academies. url: www.acola.org.au

Barnaud, C., Antona, M., Marzin, J., 2011, Vers une mise en débat des incertitudes associées à la notion de service écosystémique, *Vertigo* **11**(1).

Barreteau, O., Bots, P. W. G., Daniell, K. A., 2010, A Framework for Clarifying "Participation" in Participatory Research to Prevent its Rejection for the Wrong Reasons, *Ecology and society* **15**(2).

Barreteau, O., Bousquet, F., Attonaty, J. M., 2001, Role-playing games for opening the black box of multi-agent systems: method and lessons of its application to Senegal River Valley irrigated systems, *Jasss-the Journal of Artificial Societies and Social Simulation* **4**(2)**:**U75-U93.

Barrios, E., 2007, Soil biota, ecosystem services and land productivity, *Ecological Economics* **64**(2)**:**269-285.

Bennett, E. M., Peterson, G. D., Gordon, L. J., 2009, Understanding relationships among multiple ecosystem services, *Ecology Letters* **12**(12)**:**1394-1404.

Bluthgen, N., Dormann, C. F., Prati, D., Klaus, V. H., Kleinebecker, T., Holzel, N., Alt, F., Boch, S., Gockel, S., Hemp, A., Maller, J., Nieschulze, J., Renner, S. C., Schoning, I., Schumacher, U., Socher, S. A., Wells, K., Birkhofer, K., Buscot, F., Oelmann, Y., Rothenwohrer, C., Scherber, C., Tscharntke, T., Weiner, C. N., Fischer, M., Kalko, E. K. V., Linsenmair, K. E., Schulze, E.-D., Weisser, W. W., in press, A quantitative index of land-use intensity in grasslands: Integrating mowing, grazing and fertilization, *Basic and Applied Ecology* (0).

Boone, R. B., Galvin, K. A., BurnSilver, S. B., Thornton, P. K., Ojima, D. S., Jawson, J. R., 2011, Using Coupled Simulation Models to Link Pastoral Decision Making and Ecosystem Services, *Ecology and Society* **16**(2).

Brondizio, E. S., Gatzweiler, F. W., Zografos, C., Kumar, M., Jianchu, X., McNeely, J., Kadekodi, G. K., Martinez-Alier, J., 2010, Socio-cultural context of ecosystem and biodiversity valuation, in: *Chapter 4. The Economics of Ecosystem and Biodiversity : The Ecologicial and Economic Foundations*.

Carpenter, S. R., Mooney, H. A., Agard, J., Capistrano, D., DeFries, R. S., Diaz, S., Dietz, T., Duraiappah, A. K., Oteng-Yeboah, A., Pereira, H. M., Perrings, C., Reid, W. V., Sarukhan, J., Scholes, R. J., Whyte, A., 2009, Science for managing ecosystem services: Beyond the Millenium Ecosytem Assessment, *Proceedings of the National Academy of Science* **106**(5)**:**1305-1312.

Cocca, G., Sturaro, E., Gallo, L., Ramanzin, M., 2012, Is the abandonment of traditional livestock farming systems the main driver of mountain landscape change in Alpine areas?, *Land Use Policy* **29**(4)**:**878-886.

Cornelissen, J., Lavorel, S., Garnier, E., Diaz, S., Buchmann, N., Gurvich, D., Reich, P. B., Ter Steege, H., Morgan, H., Van Der Heijden, M., 2003, A handbook of protocols for standardised and easy measurement of plant functional traits worldwide, *Australian Journal of Botany* **51**(4)**:**335-380.

Costanza, R., 2008, Ecosystem services: Multiple classification systems are needed, *Biological Conservation* **141**(2)**:**350-352.

de Bello, F., Lavorel, S., Albert, C. H., Thuiller, W., Grigulis, K., Dolezal, J., Janecek, S., Leps, J., 2011, Quantifying the relevance of intraspecific trait variability for functional diversity, *Methods in Ecology and Evolution* **2**(2)**:**163-174.

de Bello, F., Lavorel, S., Diaz, S., Harrington, R., Cornelissen, J. H. C., Bardgett, R. D., Berg, M. P., Cipriotti, P., Feld, C. K., Hering, D., da Silva, P. M., Potts, S. G., Sandin, L., Sousa, J. P., Storkey, J., Wardle, D. A., Harrison, P. A., 2010, Towards an assessment of multiple ecosystem processes and services via functional traits, *Biodiversity and Conservation* **19**(10)**:**2873-2893.

Diaz, S., Lavorel, S., de Bello, F., Quétier, F., Grigulis, K., Robson, T. M., 2007, Incorporating plant functional diversity effects in ecosystem service assessments, *Proceedings of the National Academy of Science* **104**(52)**:**20684-20689.

Diaz, S., Quetier, F., Caceres, D. M., Trainor, S. F., Perez-Harguindeguy, N., Bret-Harte, M. S., Finegan, B., Pena-Claros, M., Poorter, L., 2011, Linking functional diversity and social actor strategies in a framework for interdisciplinary analysis of nature's benefits to society, *Proceedings of the National Academy of Sciences of the United States of America* **108**(3)**:**895-902.

Duru, M., Cruz, P., Jouany, C., Theau, J. P., 2011, Combiner des recherches en agroécologie et des dispositifs participatifs pour construire des outils d'évaluation des prairies permanentes, *Cahiers Agricultures* **20**(3)**:**223-34.

Duru, M., Felten, B., Theau, J., Martin, G., 2012a, A modelling and participatory approach for enhancing learning about adaptation of grassland-based livestock systems to climate change, *Regional Environmental Change***:**1-12.

Duru, M., Martin-Clouaire, R., 2011, Cognitive tools to support learning about farming system management: a case study in grazing systems, *Crop & Pasture Science* **62**(9)**:**790-802.

Duru, M., Tallowin, J., Cruz, P., 2005, Functional diversity in low-input grassland farming systems: characterisation, effect and management, in: *Integrating Efficient Grassland Farming and Biodiversity* (R. Lillak, R. Viiralt, A. Linke, V. Geherman, eds.), pp. 199-210.

Duru, M., Theau, J. P., Cruz, P., 2012b, Functional diversity of species-rich managed grasslands in response to fertility, defoliation and temperature, *Basic and Applied Ecology* **13**(1)**:**20-31.

Egoh, B., Reyers, B., Rouget, M., Bode, M., Richardson, D. M., 2009, Spatial congruence between biodiversity and ecosystem services in South Africa, *Biological Conservation* **142:**553-562.

Eigenbrod, F., Armsworth, P. R., Anderson, B. J., Heinemeyer, A., Gillings, S., Roy, D. B., Thomas, C. D., Gaston, K. J., 2010, The impact of proxy-based methods on mapping the distribution of ecosystem services, *Journal of Applied Ecology* **47**(2)**:**377-385.

Etienne, M., (coord), 2010, La modélisation d'accompagnement: Une démarche participative en appui au développement durable, Quae Editions, Versailles, France, pp. 368.

Fahrig, L., Baudry, J., Brotons, L., Burel, F. G., Crist, T. O., Fuller, R. J., Sirami, C., Siriwardena, G. M., Martin, J. L., 2011, Functional landscape heterogeneity and animal biodiversity in agricultural landscapes, *Ecology Letters* **14**(2)**:**101-112.

Fischer, M., Wipf, S., 2002, Effect of low-intensity grazing on the species-rich vegetation of traditionally mown subalpine meadows, *Biological Conservation* **104**(1)**:**1-11.

Fleury, P., 1997, Les prairies de fauche et de pâture des Alpes du Nord. Fiches techniques pour le diagnostic et la conduite des prairies, GIS Alpes du Nord.

Gaucherand, S., Lavorel, S., 2007, New method for rapid assessment of the functional composition of herbaceous plant communities, *Austral Ecology* **32**(8)**:**927-936.

Gellrich, M., Baur, P., Koch, B., Zimmermann, N. E., 2007, Agricultural land abandonment and natural forest re-growth in the Swiss mountains: A spatially explicit economic analysis, *Agriculture Ecosystems & Environment* **118**(1-4)**:**93-108.

Gibbons, J. M., Nicholson, E., Milner-Gulland, E. J., Jones, J. P. G., 2011, Should payments for biodiversity conservation be based on action or results?, *Journal of Applied Ecology* **48**(5)**:**1218-1226.

Griffon, S., Nespoulous, A., Cheylan, J. P., Marty, P., Auclair, D., 2011, Virtual reality for cultural landscape visualization, *Virtual Reality* **15**(4)**:**279-294.

Guyot, P., Honiden, S., 2006, Agent-Based Participatory Simulations: Merging Multi-Agent Systems and Role-Playing Games, *Jasss-the Journal of Artificial Societies and Social Simulation* **9**(4).

Hansson, B. C., Wackernagel, M., 1999, Rediscovering place and accounting space: how to re-embed the human economy, *Ecological Economics* **29**(2)**:**203-213.

Hein, L., van Koppen, K., de Groot, R. S., van Ierland, E. C., 2006, Spatial scales, stakeholders and the valuation of ecosystem services, *Ecological Economics* **57**(2)**:**209-228.

Huitric, M. E., Walker, B., Moberg, F., Österblom, H., Sandin, L., Grandin, U., Olsson, P., Bodegård, J., 2009, Biodiversity, Ecosystem Services and Resilience - Governance for a Future with Global Changes, Biodiversity, ecosystem services and governance - targets beyond 2010« on Tjärnö, Sweden, 4-6 September 2009. Albaeco, Stockholm, Sweden.

Jack, B. K., Kousky, C., Sims, K. R. E., 2008, Designing payments for ecosystem services: Lessons from previous experience with incentive-based mechanisms, *Proceedings of the National Academy of Sciences* **105**(28)**:**9465.

Kattge, J., Diaz, S., Lavorel, S., Prentice, C., Leadley, P., Bonisch, G., Garnier, E., Westoby, M., Reich, P. B., Wright, I. J., Cornelissen, J. H. C., Violle, C., Harrison, S. P., van Bodegom, P. M., Reichstein, M., Enquist, B. J., Soudzilovskaia, N. A., Ackerly, D. D., Anand, M., Atkin, O., Bahn, M., Baker, T. R., Baldocchi, D., Bekker, R., Blanco, C. C., Blonder, B., Bond, W. J., Bradstock, R., Bunker, D. E., Casanoves, F., Cavender-Bares, J., Chambers, J. Q., Chapin, F. S., Chave, J., Coomes, D., Cornwell, W. K., Craine, J. M., Dobrin, B. H., Duarte, L., Durka, W., Elser, J., Esser, G., Estiarte, M., Fagan, W. F., Fang, J., Fernandez-Mendez, F., Fidelis, A., Finegan, B., Flores, O., Ford, H., Frank, D., Freschet, G. T., Fyllas, N. M., Gallagher, R. V., Green, W. A., Gutierrez, A. G., Hickler, T., Higgins, S. I., Hodgson, J. G., Jalili, A., Jansen, S., Joly, C. A., Kerkhoff, A. J., Kirkup, D., Kitajima, K., Kleyer, M., Klotz, S., Knops, J. M. H., Kramer, K., Kuhn, I., Kurokawa, H., Laughlin, D., Lee, T. D., Leishman, M., Lens, F., Lenz, T., Lewis, S. L., Lloyd, J., Llusia, J., Louault, F., Ma, S., Mahecha, M. D., Manning, P., Massad, T., Medlyn, B. E., Messier, J., Moles, A. T., Muller, S. C., Nadrowski, K., Naeem, S., Niinemets, U., Nollert, S., Nuske, A., Ogaya, R., Oleksyn, J., Onipchenko, V. G., Onoda, Y., Ordonez, J., Overbeck, G., Ozinga, W. A., et al., 2011, TRY - a global database of plant traits, *Global Change Biology* **17**(9)**:**2905-2935.

Kemkes, R. J., Farley, J., Koliba, C. J., 2010, Determining when payments are an effective policy approach to ecosystem service provision, *Ecological Economics* **69**(11)**:**2069-2074.

Kleyer, M., Bekker, R. M., Knevel, I. C., Bakker, J. P., Thompson, K., Sonnenschein, M., Poschlod, P., Van Groenendael, J. M., Klimeš, L., Klimešová, J., Klotz, S., Rusch, G. M., Hermy, M., Adriaens, D., Boedeltje, G., Bossuyt, B., Dannemann, A., Endels, P., Götzenberger, L., Hodgson, J. G., Jackel, A. K., Kühn, I., Kunzmann, D., Ozinga, W. A., Römermann, C., Stadler, M., Schlegelmilch, J., Steendam, H. J., Tackenberg, O., Wilmann, B., Cornelissen, J. H. C., Eriksson, O., Garnier, E., Peco, B., 2008, The LEDA Traitbase: a database of life-history traits of the Northwest European flora, *Journal of Ecology* **96**(6)**:**1266-1274.

Lambin, E. F., Meyfroidt, P., 2011, Global land use change, economic globalization, and the looming land scarcity, *Proceedings of the National Academy of Sciences* **108**(9)**:**3465-3472.

Lavorel, S., Garnier, E., 2002, Predicting changes in community composition and ecosystem functionning from plant traits: revisiting the Holy Grail, *Functional Ecology* **16:**545-556.

Lavorel, S., Grigulis, K., 2012, How fundamental plant functional trait relationships scale-up to trade-offs and synergies in ecosystem services, *Journal of Ecology* **100**(1)**:**128-140.

Lavorel, S., McIntyre, S., Landsberg, J., Forbes, T. D. A., 1997, Plant functional classifications: from general groups to specific groups based on response to disturbance, *TREE* **12**(12)**:**474-478.

Le, Q. B., Park, S. J., Vlek, P. L. G., Cremers, A. B., 2008, Land-Use Dynamic Simulator (LUDAS): A multi-agent system model for simulating spatio-temporal dynamics of coupled human-landscape system. I. Structure and theoretical specification, *Ecological Informatics* **3**(2)**:**135-153.

Lopez-Hoffman, L., Varady, R. G., Flessa, K. W., Balvanera, P., 2010, Ecosystem services across borders: a framework for transboundary conservation policy, *Frontiers in Ecology and the Environment* **8**(2)**:**84-91.

Martin, G., Felten, B., Duru, M., 2011, Forage rummy: A game to support the participatory design of adapted livestock systems, *Environmental Modelling & Software* **26**(12)**:**1442-1453.

McKenna, J., Quinn, R. J., Donnelly, D. J., Cooper, J. A. G., 2008, Accurate Mental Maps as an Aspect of Local Ecological Knowledge (LEK): a Case Study from Lough Neagh, Northern Ireland, *Ecology and Society* **13**(1).

Mendoza, G. A., Prabhu, R., 2005, Combining participatory modeling and multi-criteria analysis for community-based forest management, *Forest Ecology and Management* **207**(1)**:**145-156.

Meyfroidt, P., 2012, Environmental cognitions, land change, and social-ecological feedbacks: an overview, *Journal of Land Use Science*.

Meyfroidt, P., in revision, Environmental cognitions, land change and social-ecological feedbacks: local case studies of the forest transition in Vietnam, *Human ecology*.

Mottet, A., Julien, M. P., Balent, G., Gibon, A., 2007, Agricultural land-use change and ash (Fraxinus excelsior L.) colonization in Pyrenean landscapes: an interdisciplinary case study, *Environmental Modeling & Assessment* **12**(4)**:**293-302.

Murray-Rust, D., Dendoncker, N., Dawson, T. P., Acosta-Michlik, L., Karali, E., Guillem, E., Rounsevell, M., 2011, Conceptualising the analysis of socio-ecological systems through ecosystem services and agent-based modelling, *Journal of Land Use Science* **6**(2-3)**:**83-99.

Nettier, B., Dobremez, L., Coussy, J. L., Romagny, T., 2010, Attitudes of livestock farmers and sensitivity of livestock farming systems to drought conditions in the French Alps, *Revue De Geographie Alpine-Journal of Alpine Research* **98**(1)**:**383-400.

Nettier, B., Dobremez, L., Seres, C., Pauthenet, Y., Orsini, M., Kosmala, L., Fleury, P., 2011, Biodiversity conservation by livestock farmers: advantages and shortcomings of the agri-environmental scheme 'Prairies fleuries', *Fourrages* (208)**:**283-292.

Niemeijer, D., de Groot, R. S., 2008, A conceptual framework for selecting environmental indicator sets, *Ecological Indicators* **8**(1)**:**14-25.

Plantureux, S., Ney, A., Amiaud, B., Schnyder, H., Isselstein, J., Taube, F., Auerswald, K., Schellberg, J., Wachendorf, M., Herrmann, A., 2010, Evaluation of the agronomical and environmental relevance of the CAP measure'flowering grassland', Mecke Druck und Verlag, pp. 666-668.

Quétier, 2006, Vulnérabilité des écosystèmes semi-naturels européens aux changements d'utilisations des terres, in: *Biologie des systèmes intégrés, Agronomie-Environnement*, Ecole supérieure Agronomique de Montpellier, Montpellier, pp. 269.

Quétier, F., Lavorel, S., Thuillier, W., Davies, I., 2007a, Plant-trait-based modelling assessment of ecosystem services sensitivity to land-use change, *Ecological Applications* **17**(8)**:**2377-2386.

Quétier, F., Thebault, A., Lavorel, S., 2007b, Plant traits in a state and transition framework as markers of ecosystem response to land-use change, *Ecological Monographs* **77**(1)**:**33-52.

Raymond, C. M., Bryan, B. A., MacDonald, D. H., Cast, A., Strathearn, S., Grandgirard, A., Kalivas, T., 2009, Mapping community values for natural capital and ecosystem services, *Ecological Economics* **68**(5)**:**1301-1315.

Reyers, B., O'Farrell, P. J., Cowling, R. M., Egoh, B. N., Le Maitre, D. C., Vlok, J. H. J., 2009, Ecosystem Services, Land-Cover Change, and Stakeholders: Finding a Sustainable Foothold for a Semiarid Biodiversity Hotspot, *Ecology and society* **14**(1).

Robson, T. M., Baptist, F., Clement, J. C., Lavorel, S., 2010, Land use in subalpine grasslands affects nitrogen cycling via changes in plant community and soil microbial uptake dynamics, *Journal of Ecology* **98**(1)**:**62-73.

Roling, N. G., Wagemakers, M. A. E., 1998, Facilitating sustainable agriculture: participatory learning and adaptive management in times of environmental uncertainty, Cambridge University Press, Cambridge . .

Rounsevell, M. D. A., Pedroli, B., Erb, K. H., Gramberger, M., Busck, A. G., Haberl, H., Kristensen, S., Kuemmerle, T., Lavorel, S., Lindner, M., Lotze-Campen, H., Metzger, M. J., Murray-Rust, D., Popp, A., Perez-Soba, M., Reenberge, A., Vadineanu, A., Verburg, P. H., Wolfslehner, B., 2012, Challenges for land system science, *Land Use Policy* **29**(4)**:**899-910.

Rudmann-Maurer, K., Weyand, A., Fischer, M., Stocklin, J., 2008, The role of landuse and natural determinants for grassland vegetation composition in the Swiss Alps, *Basic and Applied Ecology* **9**(5)**:**494-503.

Schwarz, G., Moxey, A., McCracken, D., Huband, S., Cummins, R., 2008, An analysis of the potential effectiveness of a Payment-by-Results approach to the delivery of environmental public goods and services supplied by Agri-Environment Schemes, Macaulay Institute, Pareto Consulting and Scottish Agricultural College. , Report to the Land Use Policy Group, UK,, pp. 108.

Schweiger, A.-K., Nopp-Mayr, U., Zohmann, M., 2012, Small-scale habitat use of black grouse (<i>Tetrao tetrix</i> L.) and rock ptarmigan (<i>Lagopus muta helvetica</i> Thienemann) in the Austrian Alps, *European Journal of Wildlife Research* **58**(1)**:**35-45.

Sommerville, M. M., Jones, J. P. G., Milner-Gulland, E. J., 2009, A Revised Conceptual Framework for Payments for Environmental Services, *Ecology and society* **14**(2).

Tappeiner, U., Tasser, E., Leitinger, G., Cernusca, A., Tappeiner, G., 2008, Effects of Historical and Likely Future Scenarios of Land Use on Above- and Belowground Vegetation Carbon Stocks of an Alpine Valley, *Ecosystems* **11**(8)**:**1383-1400.

Tasser, E., Walde, J., Tappeiner, U., Teutsch, A., Noggler, W., 2007, Land-use changes and natural reforestation in the Eastern Central Alps, *Agriculture, Ecosystems & Environment* **118**(1-4)**:**115-129.

Teich, M., Bebi, P., 2009, Evaluating the benefit of avalanche protection forest with GIS-based risk analyses-A case study in Switzerland, *Forest Ecology and Management* **257**(9)**:**1910-1919.

Tengö, M., Belfrage, K., 2004, Local management practices for dealing with change and uncertainty: a cross-scale comparison of cases in Sweden and Tanzania, *Ecology and Society* **9**(4).

Tscharntke, T., Klein, A. M., Kruess, A., Steffan-Dewenter, I., Thies, C., 2005, Landscape perspectives on agricultural intensification and biodiversity - ecosystem service management, *Ecology Letters* **8**(8)**:**857-874.

van Oudenhoven, A. P. E., Petz, K., Alkemade, R., Hein, L., de Groot, R. S., 2012, Framework for systematic indicator selection to assess effects of land management on ecosystem services, *Ecological Indicators* **21**(0)**:**110-122.

Vatn, A., 2010, An institutional analysis of payments for environmental services, *Ecological Economics* **69**(6)**:**1245-1252.

Volkery, A., Ribeiro, T., Henrichs, T., Hoogeveen, Y., 2008, Your Vision or My Model? Lessons from Participatory Land Use Scenario Development on a European Scale, *Systemic Practice and Action Research* **21**(6)**:**459-477.

Walker, B., Carpenter, S., Anderies, J., Abel, N., Cumming, G., Janssen, M., Lebel, L., Norberg, J., Peterson, G. D., Pritchard, R., 2002, Resilience Management in Social-ecological Systems: a Working Hypothesis for a Participatory Approach, *Conservation Ecology* **6**(1)**:**14.

Walz, A., Lardelli, C., Behrendt, H., Grêt-Regamey, A., Lundström, C., Kytzia, S., Bebi, P., 2007, Participatory scenario analysis for integrated regional modelling, *Landscape and Urban Planning* **81**(1-2)**:**114-131.

Willemen, L., Verburg, P. H., Hein, L., van Mensvoort, M. E. F., 2008, Spatial characterization of landscape functions, *Landscape and Urban Planning* **88:**34-43.

Wittig, B., Kemmermann, A. R. G., Zacharias, D., 2006, An indicator species approach for result-orientated subsidies of ecological services in grasslands - A study in Northwestern Germany, *Biological Conservation* **133**(2)**:**186-197.

Young, O. R., Lambin, E. F., Alcock, F., Haberl, H., Karlsson, S. I., McConnell, W. J., Myint, T., Pahl-Wostl, C., Polsky, C., Ramakrishnan, P. S., Schroeder, H., Scouvart, M., Verburg, P. H., 2006, A portfolio approach to analyzing complex human-environment interactions: Institutions and land change, *Ecology and Society* **11**(2).

Zhang, W., Ricketts, T. H., Kremen, C., Carney, K., Swinton, S. M., 2007, Ecosystem services and dis-services to agriculture, *Ecological Economics* **64:**253-260.

Annexes

Novembre 2010

Laboratoire d'écologie Alpine, Station Alpine Joseph Fourier

LA FEUILLE DU LAUTARET

Un petit bulletin pour communiquer ...

Depuis plusieurs années le Laboratoire d'Ecologie Alpine (LECA) et la Station Alpine Joseph Fourier réalisent sur l'adret de Villar d'Arène des études de recherche en écologie, complétées par des études transversales sur des thématiques agronomiques, géographiques ou sociologiques. La réalisation de ces études sont permises par la collaboration des Faranchins (habitants de Villar d'Arène) et tout particulièrement des agriculteurs de la commune. Jusqu'à présent des réunions de restitution des résultats ont été organisées chaque année. Cependant l'idée de réaliser un petit bulletin synthétisant les actions et résultats de l'année a été soulevée lors de la dernière réunion. Voici donc ce premier bulletin dont le thème principal est celui des services écosystémiques fournis par les prairies de montagne, sujet discuté lors d'entretiens réalisés en décembre 2009 et janvier 2010.

Les services écosystémiques

Le concept de services écosystémiques a été introduit dans les années 90 comme un nouveau moyen de promouvoir la conservation de la biodiversité par une approche centrée sur l'Homme, basée sur notre dépendance aux biens et services que la biodiversité nous fournit. Depuis l'*évaluation des écosystèmes pour le millénaire* (EM) (http://www.maweb.org/fr/index.aspx), achevée en 2005, les services écosystémiques ont ensuite été adoptés beaucoup plus largement au sein de différentes disciplines scientifiques et commencent à être utilisés dans le domaine politique et entrepreneurial. Ce sujet commence aussi à être diffusé dans la presse, majoritairement sous son aspect économique (monétarisation de la biodiversité). Cependant, ce concept est bien plus diversifié et demande à être mieux connu.

Les différents termes tels que services écosystémiques, écologiques, environnementaux ou du paysage, ainsi que les définitions utilisées, peuvent conduire à différentes interprétations. C'est pourquoi nous avons réalisé des enquêtes pour étudier comment ce concept est perçu. Par ailleurs, le but de ces enquêtes était de confronter la représentation de ces services par les bénéficiaires et/ou gestionnaires, à la fourniture potentielle de services par les prairies.

Les services écosystémiques sont définis par l'EM comme les bénéfices que la société obtient des écosystèmes. Des discussions autour de cette définition ont permis de soulever le fait que les services sont les bénéfices directement issus des écosystèmes (ici les prairies) et non indirectement aux travers de transformations liées aux activités humaines (par exemple : le fromage). Dans le cadre des écosystèmes agricoles, c'est une des différences majeures entre ce concept, qui peut être interprété comme une multifonctionalité des écosystèmes, et celui de la multifonctionalité de l'agriculture lié plus particulièrement aux activités agricoles. Une classification des services écosystémiques adaptée au contexte agricole permet de clarifier cet aspect (voir Figure 1 ci-dessous).

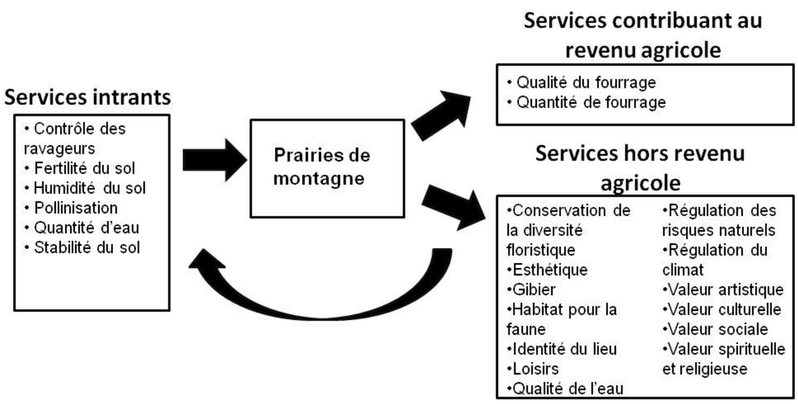


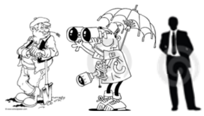
Figure 1. Classification des services écosystémiques dans le cadre des prairies de montagne. Les fonctions et processus de écosystèmes contribuent au bon fonctionnement des prairies mais aussi au revenu agricole comme au reste de la société. Source : Le Roux et al (2009) rapport disponible à l'adresse : (http://www.inra.fr/l_institut/expertise/expertises_realisees/agriculture_et_biodiversite__1)

Résultats de l'étude : les différentes représentations

Les enquêtes réalisées

Les informations synthétisées dans ce bulletin sont issues de discussions avec différentes personnes. Elles reflètent leurs représentations des prairies et des services qu'elles fournissent. Elles ont été réalisées dans trois régions de montagne de pays différents. Le canton de la Grave en France, la vallée de Stubai en Autriche et le Parc National du Yorkshire Dales en Angleterre. Ces trois régions sont dominées par une agriculture basée sur l'élevage avec des pratiques plus ou moins intensives et une activité économique touristique importante.

Premièrement, une dizaine d'entretiens individuels ont été réalisés avec des personnes travaillant dans des organismes publiques ou des associations tels que Chambre d'agriculture, Conservatoire naturel, Conseils généraux, organismes de promotion touristique. Deuxièmement, un débat avec des habitants de chaque régions a été effectué au cours de l'hiver et du printemps 2010.



Discussion de groupe à Villar d'Arène, janvier 2010 (ci-dessus) et discussion de groupe à Innsbruck (Autriche) (ci-dessous)

Les thèmes abordés

2010 est l'année internationale de la biodiversité. Mais qu'est ce que la biodiversité ?

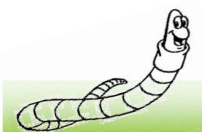
Prairie
Photo S. Aubert

La biodiversité a été unanimement décrite en terme d'espèces. La diversité des habitats a été généralement perçue, alors que les multiples échelles de la biodiversité allant du gène au paysage ont rarement été décrites.

Etonnamment, dans le cadre des prairies, ce sont surtout les espèces végétales qui ont été mentionnées, la faune étant très rarement citée. La diversité des organismes du sol n'a quasiment jamais été mentionnée. Si certains ont parlé d'espèces rares ou plutôt menacées, la diversité des espèces communes a également été mentionnée.

La biodiversité a aussi été décrite par des agriculteurs en termes d'espèces à plus ou moins larges feuilles qui conditionnent la qualité fourragère. Cette description en terme de tailles ou formes des tiges ou feuilles est appelée dans le language scientifique la diversité fonctionnelle.

Aucune différence de représentation de la biodiversité entre les trois régions étudiées n'ont été identifiées. Les différences sont plutôt liées aux connaissances individuelles.



Les **micro-organismes** (champignons et bactéries) du **sol** jouent un rôle fondamental dans le fonctionnement du sol et ils constituent de précieux indicateurs de la qualités des sols. Les échanges entre les plantes et le sol (cycle de l'azote et du carbone) jouent un rôle majeur dans la régulation de l'écosystème.

Qu'est ce que la fertilité des sols ?

Contrairement à la biodiversité, le sujet de la fertilité des sols a suscité beaucoup moins de discussion.

Dans la plupart des cas, la fertilité des sols à été expliquée en terme de fertilisation organique ou chimique. Elle a également été définie comme la capacité du sol à maintenir la croissance des plantes et la biomasse, ou encore comme la concentration en éléments minéraux et organiques. Le rôle joué par les micro-organismes n'a été mentionné que de façon anecdotique. Pourtant ceux-ci peuvent jouer un rôle important (voir encadré à gauche).

Collecte d'échantillon de sol (été 2009, Lautaret)
photo : P. Lamarque

des prairies et des services qu'elles fournissent.

Que sont les services écosystémiques ?

Contrairement à la biodiversité et la fertilité, la connaissance et la représentation du concept de services écosystémiques étaient plus contrastée entre pays. Le concept de services écosystémiques était connu d'une petite moitié des interviewés français, principalement ceux travaillant dans le domaine de la conservation, alors que ce concept est déjà plus largement adopté en Angleterre et pas du tout en Autriche. Au total 18 services ont été décrits par les interviewés dont neuf sont communs aux trois régions. Il s'agit de la stabilité du sol, la quantité d'eau, la qualité du fourrage, la conservation de la diversité floristique, l'esthétique, la valeur culturelle, la régulation des risques naturels, les loisirs, et la qualité de l'eau.

Il est important de noter que ce concept n'est pas unanimement apprécié, dû à l'aspect économique généralement mis en avant, et à son approche anthropocentrée. Par ailleurs, l'utilité de parler de services écosystémiques par rapport à d'autres concepts (ex. multifonctionalité, aménité, ressources,…) a été mis en avant.

Il semble donc qu'une meilleure communication sur les services écosystémiques, ce qu'ils recouvrent et leur intérêts est nécessaire.

Pollinisation des prairies, lieux de production d'élevage avant tout mais que les touristes aiment observer.

Différences de représentation observées

Les différences apparaissent à trois niveaux :

1) Entre pays la différence est plus marquée au niveau des habitants qu'au niveau du personnel administratif. Ceci suggère que la représentation des premiers est influencée principalement par le contexte local (invasion de campagnols, agri-tourisme, …) ou par l'usage et l'intérêt personnel. Tandis que les discours politiques (par ex. PAC et multifonctionalité) influencent et homogénéisent plus les représentations des seconds.

2) Au sein d'une même région cette dichotomie est également visible et reflète la complémentarité entre les savoirs techniques et les connaissances du terrain.

3) Les services listés suite à la définition de services sont différents de ceux considérés par les interviewés comme les plus important à préserver.

Relations perçues entre la biodiversité , la fertilité des sols et les services

La question de la relation entre la fertilité, la biodiversité et les services écosystémiques a été discutée. La figure 2 ci-contre synthétise les deux représentations qui ressortent des entretiens. D'une part, une vision de la fertilité exclusivement liée à la fertilisation qui diminue la biodiversité. D'autre part, une vision plus nuancée de l'effet combiné de la fertilisation et de facteurs tels que la pluie ou la température qui ont un effet négatif sur la biodiversité seulement dans les cas extrêmes. Dans le premier cas, seuls les services contribuant au revenu agricole sont perçus comme compatibles avec des pratiques de fertilisation, tandis que dans le second cas, à l'exception des valeurs extrêmes, l'effet décrit est positif sur de multiples services.

Une partie des expérimentations menées actuellement au Lautaret a pour but d'estimer ces effets (différentes pratiques agricoles et changements climatique) sur la végétation et la fertilité du sol.

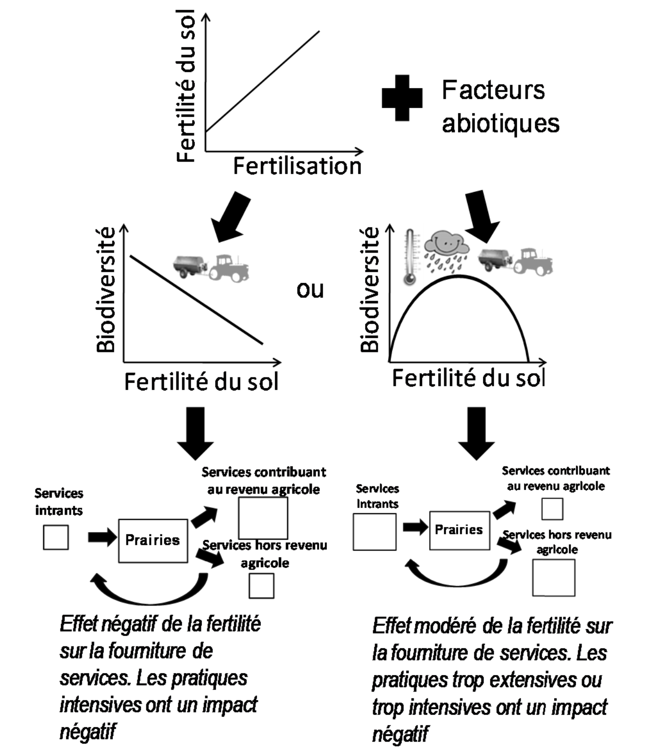


Figure 2.

Le projet de recherche VITAL

VITAL est un projet de recherche du Laboratoire d'Ecologie Alpine (financement ANR Biodiversa) qui tente de comprendre le rôle de la gestion des prairies sur la dynamique de la diversité végétale et des micro-organismes du sol, et sur les services écosystémiques qui en découlent.

Ces résultats permettront de modéliser l'évolution des services écosystémiques en fonction de différents scénarios de changement global.

L'approche participative adoptée par VITAL a pour but d'aider les gestionnaires des prairies à se préparer aux effets de ces changements et à contribuer à l' adaptation de leurs pratiques.

Par ailleurs, le projet devrait fournir des résultats intéressants pour le développement de politiques de gestion des terres en faveur de la biodiversité et du développement durable de l' élevage et des prairies.

Trois sites ont été choisis pour étudier un gradient d'intensification (voir figure 3 ci-contre).

Ce projet s'inscrit dans la continuité de projets menés sur Villar d'Arène depuis 8 ans !

Que fait-on sur le site de Villar d'Arène?

- Entretiens auprès des exploitants et des acteurs institutionnels.
- Etudes sur le terrain
- Expérimentations au jardin alpin du Lautaret
- Modélisation : cartographie, scénarios

Figure 3

Laboratoire d'Ecologie Alpine (LECA) et Station Alpine Joseph Fourrier (SAJF)

Responsable du projet VITAL : Sandra Lavorel

Laboratoire d'Ecologie Alpine, CNRS
Université J. Fourier - Grenoble I
BP 53, 2233 Rue de la Piscine

38041- GRENOBLE cedex 9

Pour plus d'information :

- **sur le projet VITAL**

http://sajf.ujf-grenoble.fr/spip.php?rubrique260

- **sur les autres projets du laboratoire et de la Station Alpine Joseph Fourrier**

http://sajf.ujf-grenoble.fr/

Et après ?

L'idée est de faire prendre conscience, au travers de cette démarche participative, des différents enjeux liés aux prairies de montagne que ce soit en terme de conservation de la biodiversité, de l'environnement ou du fonctionnement des écosystèmes agricoles. Par une approche de scénarisation, nous allons explorer comment la répartition spatiale des services peut changer selon différents scénarios futurs et discuter des moyens pour entreprendre une gestion concertée et durable des prairies. Pour ce faire, nous voudrions mettre en place un outil qui permettra d'appréhender les changements et l'importance des pratiques agricoles (par des simulations de modifications de pratiques agro-pastorales), et faire interagir les habitants sur les projets de recherche et expérimentation de terrain en cours sur leur commune. Cet outil pourra ensuite être utilisé pour la sensibilisation du public (scolaires, touristes...).

Juin 2011

Laboratoire d'écologie Alpine, Station Alpine Joseph Fourier

LA FEUILLE DU LAUTARET

Sécheresses en 2003, 2009, … et après ? Préparons nous à l'avenir !

Contexte

Les rapports du Groupe d'experts intergouvernemental sur l'évolution du climat (GIEC 2001, 2007) ont pointé les écosystèmes de montagne comme hautement vulnérables aux changements climatiques. Les scénarios d'évolution du climat prévoient non seulement une poursuite du réchauffement observé sur les Alpes, mais aussi une augmentation des extrêmes climatiques, en particulier des phénomènes de sécheresse. Les écosystèmes alpins sont considérés comme particulièrement sensibles à ces changements qui risquent d'entrainer des pertes de biodiversité et des modifications des paysages. De plus, les systèmes d'élevage des Alpes françaises sont fortement exposés au changement climatique annoncé et la plupart subissent déjà des épisodes de sécheresse. Ainsi, les adaptations des systèmes d'élevage peuvent mener à des changements de pratique agricoles qui à leur tour sont susceptibles d' atténuer ou accentuer l'évolution des écosystèmes et des services qu'ils fournissent (voir feuille du Lautaret de Novembre 2010).

Création des scénarios

Deux scénarios de sécheresses et deux scénarios socio-économiques (couplés deux à deux. Fig 1) ont été créés avec des ingénieurs et techniciens des Chambres d'Agriculture, Parc Naturel Régional, Parc National, CERPAM et ADEM afin d'être le plus pertinents possible au niveau local.

Un pas de temps de 5 ans, ce qui est relativement court pour des scénarios, a été volontairement choisi afin d'être pertinent au niveau des décisions d'une exploitation agricole.

La première étape a consisté à déterminer les principaux éléments d'une sécheresse et les principaux facteurs socio-économiques qui affectent les pratiques agricoles. Ensuite, l'effet de ces différents facteurs sur la végétation des prairies, et sur les alpages de la région du Lautaret ont été identifiés. Ceci a été réalisé dans le but de préciser et concrétiser les scénarios. Finalement des petits récits d'une vingtaine de lignes ont été rédigés afin de synthétiser et décrire les scénarios.

Les scénarios

1) «Sécheresses intermittentes» : tendance actuelle d'alternance d'années sèches (une d'été et une de printemps) et d'années normales ;

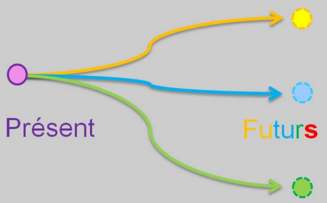
2) «Sécheresses drastiques» : quatre années successives de sécheresse de printemps ;

Dont les effets peuvent être renforcés ou atténués par:

3) «Régional» : Politiques décidées régionalement pour le maintien d'une agriculture de production mais renforcée sur les résultats en matière d'environnement; consommation locale de qualité recherchée (produits et tourisme agricole) ;

4) «International» : Politiques internationales promouvant le maintien d'une agriculture de montagne productrice de services environnementaux (ex. stockage de carbone, biodiversité). Consommation au moindre prix et tourisme sportif de montagne.

Un **scénario** est une description cohérente et plausible d'un possible état futur du monde. Mais ce n'est **pas une prévision**, au contraire, chaque scénario est une image **alternative** de comment le futur **pourrait** se dérouler.



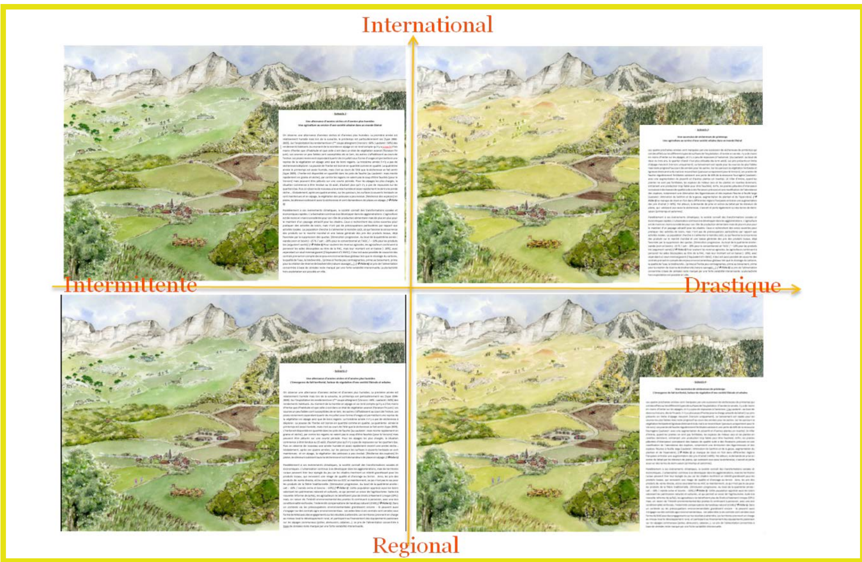


Figure 1 : Deux scénarios de sécheresses (Intermittente vs drastique) croisés avec deux scénarios socio-économique (Régional vs international)

Résultats de l'étude : Adaptation des

Une mise en situation des éleveurs à l'aide d'un jeu !

Afin d'étudier les adaptations des systèmes d'élevage, les éleveurs ont été mis en situation de sécheresse à l'aide d'un jeu. Celui-ci est composé d'un plateau représentant les différent types de prairies et alpages de leur commune. Chaque année, en fonction du climat, les joueurs doivent arriver à faire vivre leur exploitation en adaptant leurs pratiques aux ressources fourragères disponibles. Le jeu à servi de support à la discussion et à été divisé en deux parties.

Le matin, le jeu simulait le scénario drastique et l'après midi c'était au tour du scénario de sécheresse intermittente.

En fin de journée, les scénarios socio-économiques ont été discutés, mettant en avant les éléments allant à l'encontre ou favorisant les adaptations envisagées durant le jeu.

Le lendemain, par entretien individuel, les éleveurs ont pu nous soumettre leur avis sur le déroulement du jeu.

A l'issue du jeu deux types de données complémentaires sont analysés : le plateau de jeu et les discours (présenté ici) qui permettent de rendre compte du réalisme des actions réalisées durant le jeu.

Parties du jeu avec les éleveurs de Villar d'Arène, avril 2011

Scénarios « Sécheresses drastiques »

Une succession de sécheresses aussi longues n'a pas été anticipée par les éleveurs. La plupart des actions sont donc hors cadre habituel du système fourrager et les décisions sont prises en cours de campagne. Ces adaptations tactiques montrent la flexibilité du système fourrager et/ou du système d'élevage et concernent soit :

- La diminution de la part de stock de foin nécessaire: extension de la période pâturage, pâturage du regain ou des mélézins, raclage des prés d'intersaison, vente de bétail, pâturage exceptionnel des prés de fauche ;
- L'augmentation du stock de foin : fertilisation des prés de fauche, augmentation des surfaces de fauche sur le site (pâture tampon ou pente) ou fauche ailleurs ;
- Sauvegarde du pâturage d'intersaison et d'alpage: bétail laissé plus longtemps dans l'étable et achat de foin supplémentaire, vente précoce de génisses (+/- 10 jours), diminution de l'effectif du troupeau berger transhumant.
- La diversification: diminution du cheptel grâce à une meilleure valorisation par transformation des produits : viande ou fromage;
- L'arrêt de l'exploitation.

Néanmoins certaines contraintes empêchent la réalisation de certaines adaptations (voir « les freins »).

En terme d'utilisation du sol, les principaux changements sont l'abandon total de la fauche après 2 ans de sécheresse, l'augmentation du chargement des prés et alpages et la fertilisation des prés de fauche et certaines pâtures (terrasses principalement).

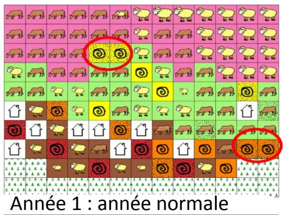
Il est intéressant de noter que même si le besoin de stratégies et décisions collectives a été mentionné par certains durant le jeu, aucune action collective n'a été réalisée à l'exception d'échange de parcelles entre éleveurs. Toutefois, certaines actions réalisées individuellement devrait obligatoirement passer par une décision collective dans la réalité (modification règlement AFP, contrat du berger).

Scénarios « Sécheresses intermittentes »

A l'inverse du scénario précédant, celui-ci est considéré par les éleveurs comme leur quotidien depuis 2003. Dès lors, les conséquences ont déjà été plus ou moins intégrées dans la gestion de l'exploitation selon les éleveurs. Néanmoins, il s'agit principalement d'actions au « coup par coup » de gestion des surfaces fourragères et des stocks. La principale différence avec le scénarios ci-dessus est la mise en place de stock de foin les années productives, notamment en fauchant de manière intermittente certains prés y compris l'année de sécheresse printanière.

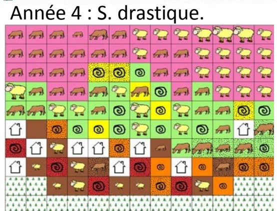
Selon les éleveurs, dans ce type de scénarios la problème se pose surtout sur les prés de fauche et dans une moindre mesure sur les prés d'intersaison. Les alpages sont surdimensionnés (marge tampon) afin de sécuriser la ressource en cas de sécheresse ou autres aléas.

A droite : Evolution du plateau de jeu selon les scénarios. Année 4, sécheresse de printemps



Année 1 : année normale

Année 4 : S. drastique.



Année 4 : S Intermittentes

pratiques agricoles à différents scénarios

Scénarios socio-économiques qui régissent l'activité agricole

A la fois source d'incertitudes et d'inquiétudes supplémentaires pour l'éleveur, les éléments des deux scénarios socio-économiques ont été discutés afin d'évaluer comment ils viennent contrarier ou aider l'exploitation à s'adapter aux sécheresses.

Selon les éleveurs l'effet de ces changements (surtout politiques) pourraient être plus dramatiques que la sécheresse car les primes constituent un apport financier essentiel au fonctionnement de l'exploitation. Pour le moment aucune adaptation n'a été proposées pour pallier à ce déficit financier qui alourdit le bilan du à l'achat de foin dans le scénarios « choc ».

Concernant les subventions, le changement majeur qui pourrait s'opposer à l'arrêt de la fauche concerne l'arrêt de primes liées au chargement (raison pour laquelle la diminution du troupeau n'a jamais été évoquée). Celles-ci pourrait être remplacées par un système de prime à l'hectare, à la fauche (autres que les MAE qui ne compensent pas selon les éleveurs le coût du maintien de cette pratique en cas de sécheresse) ou via un revenu minimal garanti. Des mesures renforcées par rapport à des résultats en matière de biodiversité ne leur semblent pas contraignants sur leur site.

L'aide en faveur d'équipements pastoraux notamment en rapport avec la gestion de l'eau en alpage (abreuvoir, retenue collinaire) serait intéressante à étudier car selon certains d'entres eux lors d'une sécheresse certaines zones de l'alpage sont moins utilisable. La création de pistes pourrait permettre de refaucher des prés difficilement accessibles (surtout à La Grave).

Une augmentation de la demande pour des produits locaux renforcerait la vente de produits transformés (viande pour locaux et restaurant) mise en place par certains agriculteurs et permettrait à d'autres éleveurs de se diversifier, à condition toutefois de diversifier l'offre. Ceci permettrait, grâce à la plus-value, de diminuer l'effectif du troupeau. Cependant, ils ne pensent pas réaliste de pouvoir profiter de l'attrait touristique du site (saison touristique courte et touriste de passage) notamment pour diversifier par de l'accueil ou de l'hébergement à la ferme d'autant plus que les bâtiments et le fonctionnement de l'exploitation ne le permet pas. Selon eux, le tourisme serait plutôt profitable dans le cadre du scénario international où les activités de loisirs tels que les stations de ski proposent une nombre important d'emplois (pour pluriactivité ou en cas de cessation d'activité).

Finalement, l'urbanisation n'est pas une menace réelle dans le secteur (non urbanisable-zone rouge), même si des prés de fauche isolés peuvent petit à petit être construits.

Freins à l'adaptation

Plusieurs adaptations discutées durant le jeu se heurterait dans la réalité à différentes contraintes. Celles-ci sont d'ordre technique (caractéristique au fonctionnement de l'exploitation), stratégique (contradiction avec le projet d'exploitation) ou territorial (ressources naturelles ou foncières non disponibles) cette dernière étant probablement le principal frein sur le Canton de la Grave.

Les conditions climatiques rendent la durée d'hivernage extrêmement longue (7-8 mois) et les exploitations très dépendantes du stock de foin et des compléments alimentaires. Hors, les terrains escarpés des prairies de haute montagne sont difficiles d'accès (pente, chemins ou route), en plus d'être parfois loin de l'exploitation (temps de travail et coût de déplacement élevé). Par conséquent, certaines prairies ne peuvent plus ou difficilement être fauchées ou fertilisées avec le matériel actuel. Les faible disponibilités foncières permettent rarement d'aller faucher ailleurs. Cependant, du terrain va se libérer avec le départ en retraite d'agriculteurs. Mais vaut il mieux le répartir entre éleveurs ou essayer d'installer un jeune agriculteur malgré les bâtiments agricoles difficilement reprenables en l'état ?

Ce sont ces mêmes bâtiments agricoles et la fragilité économique des exploitations rendant difficile tout investissement supplémentaire qui freinent en partie les éleveurs à se diversifier (atelier de traite ou fabrication, visite de ferme, …) ou à faire certains choix d'adaptation. Mais d'autres facteurs entrent également en compte tels que la vision du métier (éleveur allaitant différent de laitier, de fromager, de berger ou d'alpagiste), les projets d'exploitation et les ressources humaines (les éleveurs sont généralement seuls sur l'exploitation et se font aider ponctuellement par la famille).

La situation économique et le fonctionnement des exploitations remet aussi en cause l'arrêt total de la fauche dans le cas où un rendement suffisant de fourrage ne pourrait être assuré durant plusieurs années. Est-il plus rentable d'acheter la totalité du foin et de revendre le matériel de fauche dont l'amortissement et la maintenance coûte

Fauche à la motofaucheuse des prairies en pente . Photo: E. Deboeuf

cher ou de continuer à faucher et acheter le foin manquant, sachant qu'actuellement la majorité des exploitations ne peut pas assumer plus de deux années d'achat complet de foin? Mais là aussi, les arguments économiques ne sont pas les seuls facteurs influençant les décisions prises par les éleveurs. Ainsi, leur attitudes, valeurs ou perceptions face au aléas et aux changements induits par les adaptations proposées pèsent dans le poids de l'arrêt de la fauche. Le respect des terrains et notamment des terrasses façonnées par les générations précédentes à été cité à plusieurs reprises.

Les projets de recherche VITAL et SECALP

VITAL est un projet de recherche du Laboratoire d'Ecologie Alpine (financement ANR Biodiversa) qui tente de comprendre le rôle de la gestion des prairies sur la dynamique de la diversité végétale et des micro-organismes du sol, et sur les services écosystémiques qui en découlent.

Ces résultats permettront de modéliser l'évolution des services écosystémiques en fonction de différents scénarios de changement global.

Ce projet s'intéresse à trois site dont le Lautaret en France et deux autres sites en Autriche et en Angleterre.

L'approche participative adoptée par VITAL a pour but d'aider les gestionnaires des prairies à se préparer aux effets de ces changements et à contribuer à l' adaptation de leurs pratiques.

SECALP est un projet de recherche du Laboratoire d'Ecologie Alpine et du Cemagref de Grenoble (financement GICC-MEDDTL) étudiant l' adaptation des territoires alpins à la recrudescence des sécheresses dans un contexte de changement global. Il vise plus particulièrement à produire des connaissances pour appuyer les acteurs, les gestionnaires et les décideurs des territoires alpins à la réduction des impacts et l'adaptation à la récurrence des sécheresses, dans un contexte de changements du climat et de la conjoncture politique et socio-économique.

Ces projets s'inscrivent dans la continuité de projets menés sur Villar d'Arène depuis 2003 !

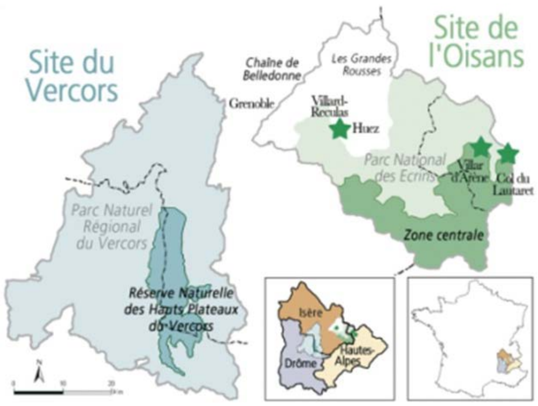


Figure : Sites d'étude du projet SECALP.

Laboratoire d'Ecologie Alpine (LECA) et Station Alpine Joseph Fourrier (SAJF)

Responsable du projet VITAL et du projet SECALP : Sandra Lavorel

Laboratoire d'Ecologie Alpine, CNRS
Université J. Fourier - Grenoble I
BP 53, 2233 Rue de la Piscine

38041- GRENOBLE cedex 9

Pour plus d'information :

- **sur le projet VITAL**

http://sajf.ujf-grenoble.fr/spip.php?rubrique260

- **sur les autres projets du laboratoire et de la Station Alpine Joseph Fourrier**

http://sajf.ujf-grenoble.fr/

Et après ?

Les adaptations décrites nous permettent d'élaborer des scénarios d'évolution de l'agriculture et des effets sur le paysage, la biodiversité et les services écosystémiques. L'évaluation des effets des changement de pratiques est réalisé sur base des expériences menées au Lautaret, de discussions avec des experts et d'expériences menées ailleurs (bibliographie).

Ces différents changements seront présentés aux touristes cet été lors d'enquêtes, avant d'être l'objet d'un nouveau jeu (à l'automne prochain) au cours duquel les éleveurs pourront observer l'effet de leur pratiques à long terme(en 2030).

Finalement, nous modéliserons à l'échelle du Canton de La Grave, l'évolution des services écosystémiques (voir feuille du Lautaret de Novembre 2010 pour une définition de ce concept).

Summary

The ecosystem service (ES) concept is increasingly used in different scientific disciplines and is spreading into policy and business circles to draw attention to the benefits that people receive from biodiversity and ecosystems. Nevertheless, while the number of case studies considering various dimensions of the interactions between ecosystems and land use via ES has been steadily increasing, integrated research addressing interrelationships between biodiversity, ES and land use has remained mostly theoretical.

This thesis aims through a socio-ecological approach to understand: (1) Which ES are potentially delivered given ecological dynamics, (2) how these ES are perceived by stakeholders in terms of value and knowledge, (3) how human management affects ES delivery, and (4) how ES are taken into account in land management decisions, thereby considering feedbacks from ecosystem to the land use system through ES.

To address these questions, a transdisciplinary study was conducted on Villar d'Arène (French Alps) a municipality where the subalpine landscape is shaped by extensive mountain livestock farming. Statistical modelling and geographical information systems where combined to analyse the determinants of the spatial distribution of biodiversity and ES within the landscape using ecological (including plant functional traits), biophysical and land-use data. The following ES were mapped: agronomic value, aesthetic value, water quality, carbon storage, soil fertility, soil moisture, conservation of plant diversity and pollination. These allowed us to quantify trade-offs and synergies in the current landscape and to identify key management types supporting multifunctionality. The dynamics of ES was projected under four different scenarios integrating climatic, socio-economic and land-use changes, which were developed using a participative approach with regional experts and local farmers. Analyses of projected scenario impacts showed that ES synergies and trade-offs evolve differently when considering direct effects of climate on ecosystems, and/or their indirect effects through farmers adaptive responses. Interviews with local stakeholders (experts from nature conservation and agricultural extension, farmers and inhabitants) of mountain grasslands showed that the ES concept is still relatively unknown in explicit terms. Nevertheless after defining ES to interviewees, they expressed a variety of relevant interests and knowledge. Although all stakeholders valued a common set of ecosystem services (agronomic value, aesthetic value, water quality, and conservation of plant diversity), we identified negative and positive representations of the effects of grassland management on ecosystem services, depending on stakeholders perceptions of the relationships between soil fertility and biodiversity, and biodiversity and the other services. Finally, a role-playing game explored how ES cognition mediated environmental feedbacks on farmers' behaviours. Results emphasized the influence of other factors such as socio-economic or climatic context, topographic constraints, social value of farming or farmer individual and household characteristics, on the link between ES and land-management decisions. This case study demonstrates the interest of an integrated approach decomposing the feedback loop from ecosystems to land use when studying ES for scientific or policy purposes.

Résumé

Le concept de services écosystémiques est de plus en plus utilisé par différentes disciplines scientifiques et pris en compte dans les sphères politiques pour attirer l'attention sur les bénéfices que l'Homme reçoit des écosystèmes. Ce concept mène à étudier les liens complexes entre l'homme et son environnement. Cependant, la majorité des recherches actuelles reste théorique et peu de cas d'étude mettent à l'épreuve ce concept dans une démarche transdisciplinaire. Cette thèse a donc pour objectif principal de combler ce manque en explorant et analysant les dynamiques et processus des services écosystémiques en terme d'offre et de demande, y compris les effets de rétroactions de changements de services sur la prise de décisions des acteurs, par une approche socio-écologique dans un contexte de changement planétaire. A cette fin, une étude a été conduite sur les prairies subalpines de la commune de Villar d'Arène (Hautes-Alpes) où l'élevage ovin et bovin domine.
Une analyse spatiale de la fourniture potentielle de multiples services écosystémiques (fertilité du sol, stockage de carbone, qualité de l'eau, quantité et qualité du fourrage, conservation de la diversité floristique et esthétique) et leurs interrelations (synergies et compromis) a été réalisée à l'échelle du paysage à partir des données de traits fonctionnels, de propriétés du sol et de pratiques agricoles. La perception et la demande en services écosystémiques ont quant à elles été étudiées par une démarche participative auprès des acteurs locaux et des agriculteurs montrant l'appropriation de ce concept par les différents acteurs ainsi que les connaissances sur les services et leurs classements par ordre d'importance. Finalement, la co-construction avec les acteurs locaux de scénarios climatiques et socio-économiques à l'horizon 2030 a permis d'étudier leurs effets sur l'utilisation du sol et de la végétation, afin d'explorer l'évolution de l'offre et la demande en services et les effets de rétroactions sur la prise de décision des agriculteurs. Les compromis entre services écosystémiques fournis dans les différents scénarios montrent comment ils sont pris en compte dans l'arbitrage des décisions de gestion des agriculteurs, malgré le rôle prépondérant d'autres facteurs (ex. politique, climat, topographie).