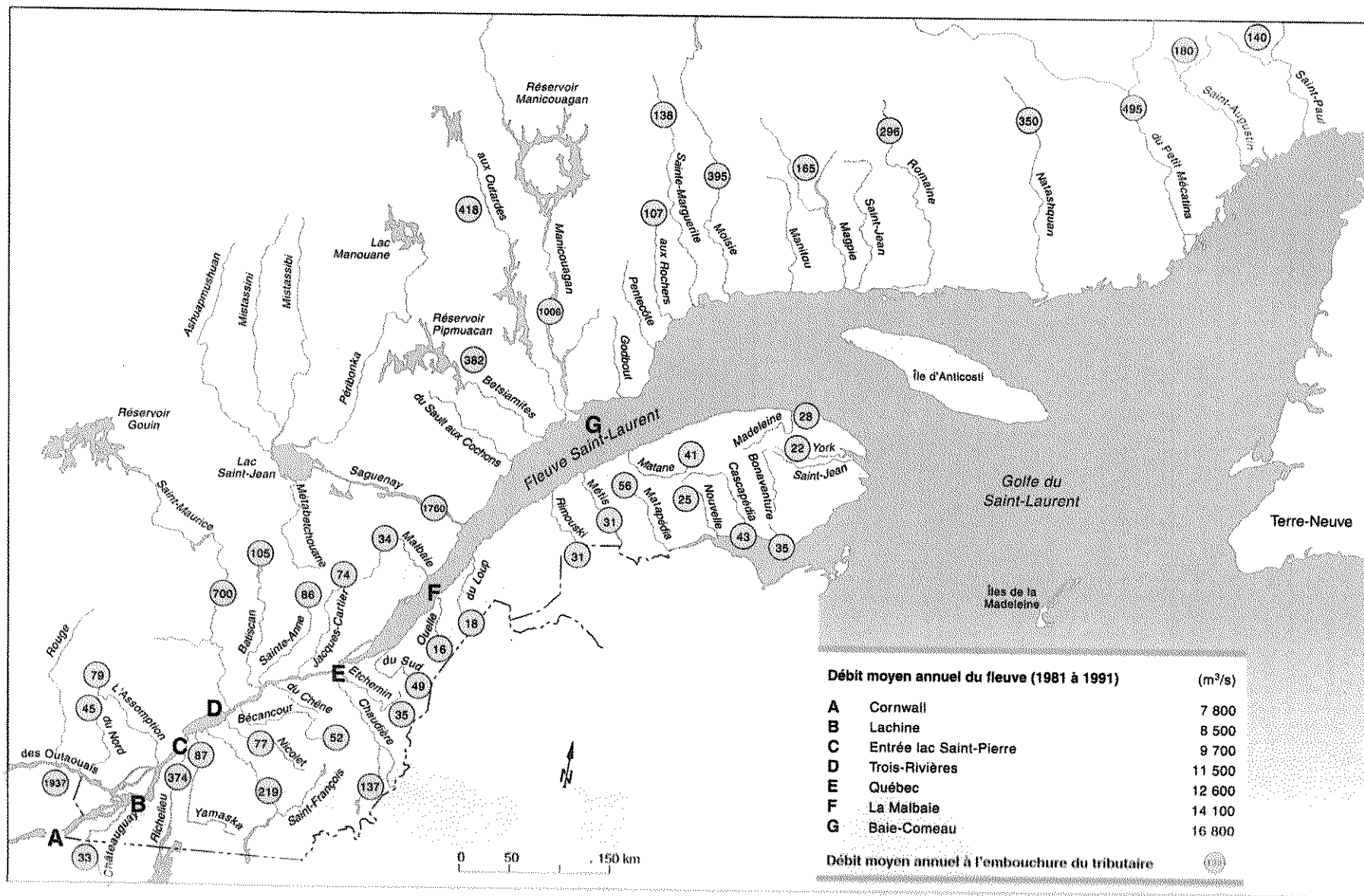


FIGURE 1.10
Débit moyen annuel du Saint-Laurent entre Cornwall et Baie-Comeau





Garrot d'Islande photographié dans l'archipel de Mingan printemps 2008



Débris de végétaux sur la grève automne 2008



Les quantités peuvent être importantes de cette matière végétale émanant des rivières.

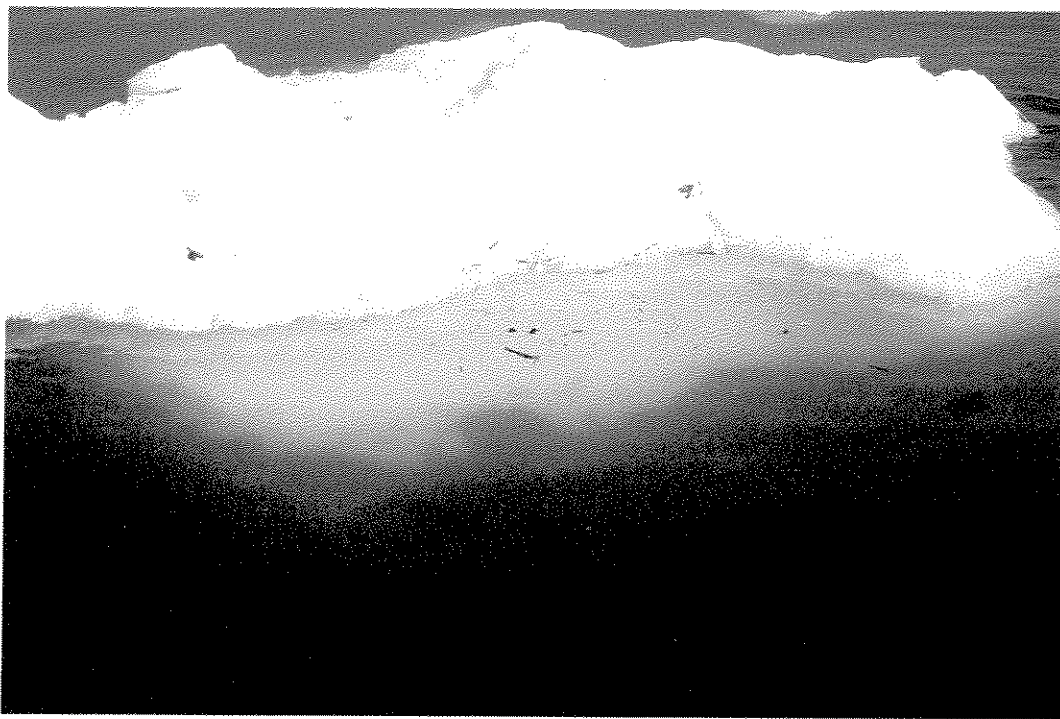
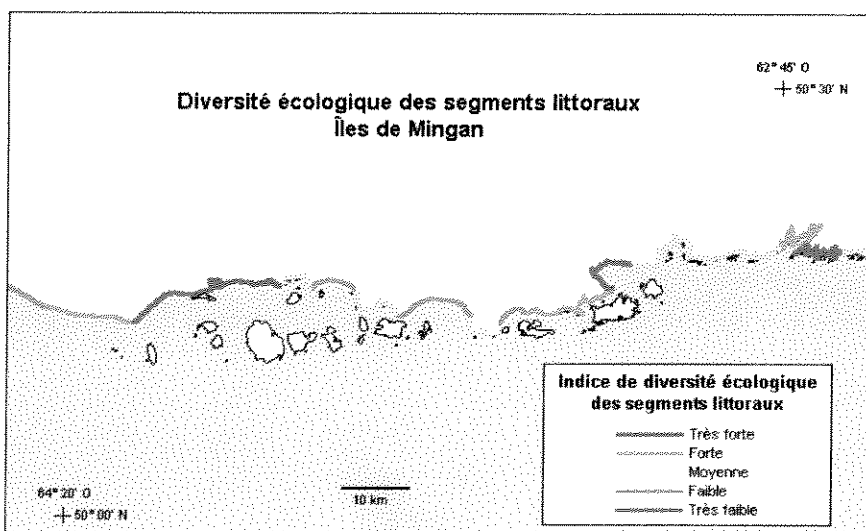
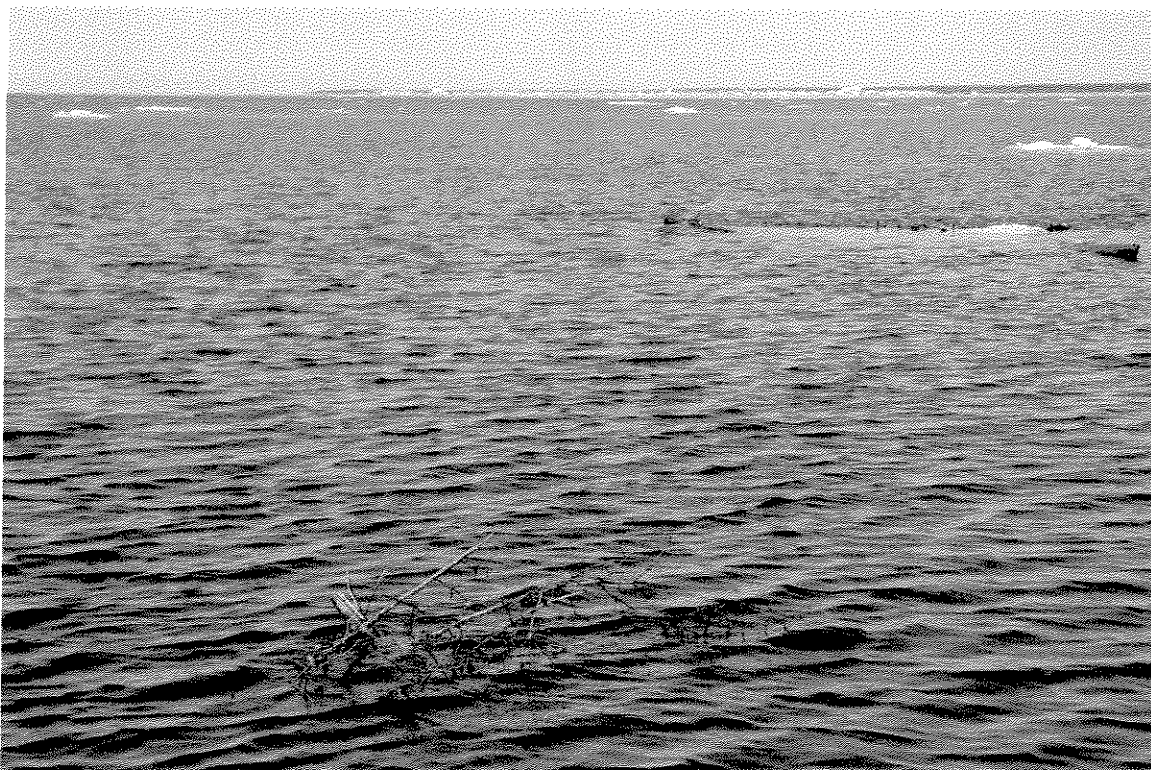


Photo avril 2008 en face de Mingan. On observe la turbidité relié a la matière organique colloïdale en suspension seulement a la débâcle.



Pi



Caractérisation de
l'importance
écologique des
différents sites
candidats à protéger
dans l'archipel
Mingan, définis par
Environnement
Canada



English

Accueil

Propos du PAN

Les Regions

Jeux

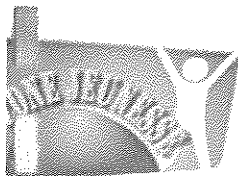
Publications

Recherche

Contactez-nous

Recherche

Recherche avancée



Protection des milieux côtiers et marins du Canada

Précédent : Table des matières : Suivant

Les enjeux clés

Modification et destruction de l'habitat

Ces dernières années, le Canada a progressivement adopté des mesures aux niveaux local, provincial, territorial, national et international pour lutter contre la modification et la destruction de l'habitat par l'application de divers programmes, politiques et mesures législatives. Ce volet est assez vaste et a été divisé en six catégories, mais bon nombre d'activités, de programmes ou de mesures législatives de protection et de conservation de l'habitat s'appliquent à plus d'une de ces catégories.

à l'échelle nationale, certaines dispositions de la protection de l'habitat de la *Loi sur les pêches*, administrées par Pêches et Océans Canada, interdisent tout projet ou activité entraînant la détérioration, la destruction ou la perturbation de l'habitat du poisson. La *Politique de gestion de l'habitat du poisson*, à l'appui de cette loi, donne un cadre détaillé à la conservation, à la restauration et à la mise en valeur de l'habitat du poisson.

En 1997, le Canada s'est engagé, par sa *Loi sur les océans*, à une gestion et une gouvernance modernes des océans. Cette loi est fondée sur le développement durable, la gestion intégrée et l'approche de précaution. Son application est facilitée par certaines mesures comme les initiatives de la gestion intégrée, les zones de protection marine et les programmes de la qualité du milieu marin.

La *Loi canadienne sur l'évaluation environnementale* a pour objet de garantir que tous les projets financés ou régis par le gouvernement fédéral n'aient pas d'incidences nuisibles appréciables sur l'environnement.

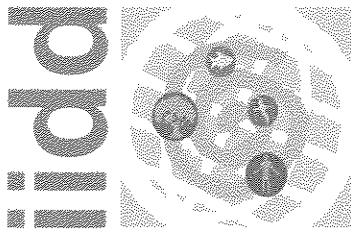
Les matériaux synthétiques ne se dégradent pas rapidement et peuvent rester dans le milieu marin pendant plusieurs années.

En octobre 1996, les ministres fédéral, provinciaux et territoriaux responsables de la faune signaient un *Accord pour la protection des espèces en péril*. Cet Accord énonce les principes de base de la conservation des espèces et fait état de divers engagements à l'égard de la protection des espèces en péril et de leur habitat. La *Loi sur les espèces en péril* s'insère dans la stratégie adoptée par le gouvernement du Canada pour respecter ses

engagements en vertu de l'Accord. La loi favorise le rétablissement des espèces en péril et la protection des habitats qui leur sont essentiels.

Les provinces et les territoires adoptent des mesures législatives qui instituent la planification intégrée à titre de fondement des décisions réglementaires et d'encadrement des activités de conservation et de protection. Le Conseil de gestion des ressources fauniques du Nunavut est un exemple de cette démarche. Le Conseil a pour mandat d'assurer la protection et l'utilisation prudente de la faune et de son habitat pour le bénéfice à long terme des Inuits et des autres résidents du Canada. Le Conseil donne des avis et formule des recommandations aux gouvernements sur les zones marines voisines et approuve les plans de gestion et de protection d'espèces et d'habitats particuliers. D'autres conseils de gestion répartis dans tout le pays assument un rôle semblable.

On trouve dans les sections qui suivent la description des six catégories du volet du PAN sur la modification et la destruction de l'habitat ainsi que des exemples d'initiatives en cours.



LES FAITS MARQUANTS DE LA CONFÉRENCE INTERNATIONALE "BIODIVERSITÉ: SCIENCE ET GOUVERNANCE": MARDI 25 JANVIER 2005

Mardi, les participants à la Conférence Internationale "Biodiversité: Science et Gouvernance" se sont réunis en Plénière tout le long de la journée. Ils ont planché sur la question de l'état actuel et de l'évolution de la biodiversité mondiale, le matin, et sur celle des avantages écologiques et sociaux de la biodiversité, l'après-midi.

PLÉNIÈRE

ÉTAT ACTUEL ET ÉVOLUTION DE LA BIODIVERSITÉ MONDIALE: Mary Arroyo, de l'Université du Chili, a présidé la session plénière consacrée à la question de l'état actuel et de l'évolution de la biodiversité mondiale.

Georgina Mace, de la Société Zoologique de Londres, a présenté l'audit mondial de l'état actuel de la biodiversité, établi sur la base de trois études: l'Évaluation Écosystémique du Millénaire, l'Évaluation Mondiale des Espèces, de l'UICN – de l'Union Internationale pour la Conservation de la Nature, et l'Eco-régions de la WWF. S'agissant du niveau intra-espèces, elle a précisé que les renseignements sont épars, que les données disponibles ne portent uniquement que sur les espèces essentielles à l'alimentation et à l'agriculture, et que les renseignements concernant les populations sont limités aux échelles régionales et aux vertébrés. Au sujet du niveau des espèces, elle a noté que les renseignements et les données disponibles sont limités aux grands groupes d'espèces, et que la diversité des espèces se situe entre 5 et 30 millions, dont 2 millions seulement sont identifiés. Elle a souligné que l'estimation du taux d'extinction des différentes espèces se situe entre 1 et 12 millions d'espèces par an, et que, selon la Liste Rouge établie par l'UICN, 12 à 52 % des espèces appartenant aux taxons élevés, sont menacées d'extinction. Au sujet des écosystèmes, elle a expliqué que l'érosion des habitats a été constante et qu'elle se poursuivrait. Mace a mis également en lumière: les lacunes importantes dans les connaissances concernant, en particulier, certains taxons importants, comme les invertébrés, les plantes et les champignons; la nécessité de procéder à l'étude de l'évolution de la biodiversité sur la base de mesures comparable sur des périodes de temps données; le manque d'études au niveau génétique; la connaissance moindre des habitats des milieux marins et d'eaux douces par rapport aux habitats terrestres; et la mauvaise compréhension des effets des changements survenus dans la biodiversité, sur les services écosystémiques.

Michael Donoghue, de l'Université de Yale, a présenté les défis à relever dans les domaines de la documentation et de la classification de la biodiversité. Il a décrit le rôle de la croissance démographique dans la destruction des habitats et l'homogénéisation des biotopes. Il a identifié les problèmes posés à l'estimation des nombres des espèces, y compris des espèces cryptiques, des différentes populations au sein des espèces et des ressources affectées à la recherche portant sur des taxons particuliers. Il a attiré l'attention sur le manque de connaissances concernant le nombre d'espèces marines, en particulier, les organismes bactériens marins. Au sujet de la relation phylogénique existant entre les espèces, il a présenté des exemples de rapports évolutifs inattendus entre diverses espèces. Il a également mis en relief l'importance de l'extraction de données des 2,6 milliards de spécimens du monde entier, qui se trouvent dans les musées, et a souligné que l'éventail des outils d'identification des

espèces s'est élargi d'année en année mais que bon nombre d'outils de laboratoire sont encore inaccessibles aux pays en développement. Donoghue a également souligné l'importance de la mise à contribution de la technologie et de l'accroissement des ressources humaines et du renforcement des capacités dans le domaine de l'identification des espèces. Il a affirmé que les outils actuellement disponibles sont en mesure de révolutionner l'identification des espèces, pour peu qu'ils soient assortis d'engagements financiers et politiques.

Achim Steiner, Directeur Général de l'UICN, a suggéré de ~~plutôt expliquer pourquoi la biodiversité est importante que de se focaliser sur l'extinction et sur la biodiversité non encore découverte.~~ Faisant constater une augmentation des zones de la Planète devenues aires protégées, il a fait part de sa satisfaction quant aux réussites enregistrées récemment dans le domaine de la préservation de la biodiversité. Il a souligné la nécessité d'unifier les approches et les stratégies et de déterminer des objectifs communs, notant que l'approche écosystémique fournit un parapluie sous lequel la recherche, la gestion et la préservation peuvent être alignées. Steiner a encouragé une participation accrue du Sud dans la préservation, et a déclaré que la nature ne devrait pas être assujettie uniquement aux analyses économiques du rapport coût-bénéfice. Faisant l'éloge des initiatives telles que la Liste Rouge de l'UICN et l'Évaluation Mondiale des Espèces, il a mis en garde que le problème n'est pas tant celui de l'élaboration des preuves élémentaires que de convaincre la société à opérer des changements. Plutôt qu'un service d'information centralisé, il a plaidé pour l'amélioration de l'interface et de la connectivité entre les différentes sources disponibles, en particulier dans les pays en développement. Steiner a suggéré de communiquer que la biodiversité est essentielle à l'éradication de la pauvreté et, mettant en relief le pouvoir du marché, a déclaré que le commerce peut être une solution et non un problème.

Le Président de la Table Ronde Jose Sarukhan, de l'Institut Ecologique National du Mexique, a invité les panélistes à échanger leurs vues concernant le rôle des chercheurs et des naturalistes amateurs et concernant le point de savoir si un centre international spécialisé en biodiversité était nécessaire.

Peter Bridgewater, Secrétaire Général de la Convention de Ramsar sur les Terres Humides, a appelé à une focalisation sur le rôle de la biodiversité dans la fourniture des services écosystémiques, traitant également des zones froides et des zones humides de l'habitat de la biodiversité, et de l'amélioration des synergies pour les avis scientifiques. Il a souligné la nécessité d'une gestion adaptative et d'une approche préventive.

Thomas Lovejoy, Président du Centre Heinz, a souligné la nécessité de déterminer les voies et moyens de montrer les liens existant entre la biodiversité et d'autres questions, y compris l'étude de la réactivité de la nature face aux changements climatiques.

Catherine Day, Directrice Générale de l'Environnement au sein de la Commission Européenne (CE), a axé son intervention sur les besoins des décideurs politiques, soulignant la nécessité de davantage de science, de meilleurs outils de mesure, et d'indicateurs politiquement pertinents. Elle a précisé que les défis à relever englobent l'intégration de la biodiversité dans les autres politiques et la persuasion des décideurs politiques que les mesures de préservation de la biodiversité, y compris l'établissement d'aires protégées, constituent un investissement pour l'avenir.

Bertrand Tramier, Directeur Exécutif de la 'Total Corporate Foundation', a souligné le besoin d'une meilleure compréhension

des liens entre l'industrie et la préservation de la biodiversité, et a mis en relief les activités menées par Total dans le domaine de la biodiversité, notamment la création d'une fondation pour la préservation de la biodiversité et des mers.

Mace, appuyée par Lovejoy, a déclaré que les aires protégées ne sont pas l'unique réponse pouvant être apportée à l'érosion de la biodiversité, et que les êtres humains doivent apprendre à vivre de manière durable avec la nature. Elle a encouragé les scientifiques à collaborer aux projets de grande échelle visant à concrétiser l'objectif 2010.

Dans le débat qui a suivi, les participants ont souligné le besoin: de dialogues ouverts entre le commerce et l'industrie, les gouvernements, les organisations internationales et les scientifiques, sur l'utilisation du bois d'œuvre certifié; d'un soutien accru de la part des gouvernements, en faveur des communautés locales, pour une utilisation durable de la biodiversité; d'un partage des avantages; de partenariats entre le Nord et le Sud, axés sur la recherche scientifique et la formation; et d'approches interdisciplinaires.

AVANTAGES ECOLOGIQUES ET SOCIAUX DE LA BIODIVERSITE: Jacques Weber, Directeur de l'Institut Français de la Biodiversité, a présidé la session plénière consacrée au thème des avantages écologiques et sociaux de la biodiversité.

David Tilman, de l'Université de Minnesota, a présenté le travail de recherche mené sur le lien entre les services écosystémiques et la biodiversité. Il a donné un aperçu historique sur les premières théories écologiques qui associaient la haute productivité à des domaines élevés de la biodiversité. Il a également présenté les résultats de la recherche menée au cours de la décennie écoulée, confirmant ces premières théories et démontrant que l'érosion de la biodiversité conduit à un déclin: de la productivité végétale; de l'efficacité de l'utilisation des ressources végétales; de la stabilité et de prévisibilité des écosystèmes; et de la fixation du carbone. Tilman a également présenté des travaux de recherche montrant que l'érosion de la biodiversité peut aboutir à des risques accrus d'invasions par des espèces exotiques. Il a précisé que les services écosystémiques englobent la productivité, la qualité de l'eau, la stabilité des écosystèmes et la protection contre les espèces exotiques envahissantes. Soulignant que la vraie valeur des services écosystémiques est souvent négligée, il a souligné qu'une politique environnementale sage est celle qui s'attelle à optimiser à la fois la quantité et la qualité de ces services au profit de la société. Dans le débat qui a suivi, les participants ont appelé à un accroissement des pratiques écologiques dans les politiques agricoles et à l'identification d'alternatives viables pour les agriculteurs qui utilisent des terres riches en biodiversité, à des fins agricoles, dans les pays en développement.

Andrew Dobson, de l'Université de Princeton, a traité des liens entre biodiversité et santé humaine. Il a affirmé que de nombreux pathogènes ne sont aptes à menacer les êtres humains que lorsque leur environnement naturel a subi des perturbations, et a argué que les solutions rudimentaires apportées aux maladies sont souvent les plus efficaces. Soulignant l'importance de la mise en lumière des connaissances écologiques, il a décrit les interactions entre les pathogènes, les systèmes immunitaires des êtres humains, la résistance aux médicaments, la densité démographique, la densité du cheptel, la biodiversité et le climat. Dobson a précisé que les espèces exotiques deviennent envahissantes souvent parce qu'elles ne trouvent pas les parasites qui les infestent dans leur habitat naturel, et que les espèces exotiques peuvent introduire des parasites capables de menacer d'extinction les espèces autochtones. Il a expliqué que l'élimination des pathogènes des systèmes naturels, comme cela a été fait à des fins agricoles dans certains endroits, influence grandement le fonctionnement d'un écosystème. Il a souligné que la biodiversité est une importante barrière contre les maladies et, en particulier, contre les maladies transmises par des vecteurs comme le paludisme, car les vecteurs tendent à s'attaquer aux êtres humains lorsque la biodiversité s'affaiblit. Il a conclu que la description des chaînes alimentaires et de la toile des écosystèmes en termes mathématiques constitue le plus grand défi scientifique à relever au 21^e siècle.

Charles Perrings, de l'Université de York, a présenté un exposé sur l'économie et la valeur de biodiversité et des services écosystémiques. Il a expliqué que la valeur anthropocentrique de la biodiversité et des écosystèmes dérive de la valeur des biens et services qu'ils fournissent, laquelle englobe des valeurs

d'utilisation directe, des valeurs d'utilisation indirecte, et des valeurs non utilitaires ou passives. Il a fait observer que la biodiversité soutient le fonctionnement et les processus des écosystèmes, qui, à leur tour, soutiennent la production des biens et services commercialisables. Il a précisé, qu'en termes économiques, la biodiversité est équivalente à un portefeuille d'actifs, et que les efforts de préservation devraient être désignés en fonction des moyens et du rapport de covariance du portefeuille. Perrings a précisé que l'économie devrait être intégrée dans les efforts et les plans de préservation, et a suggéré de procéder à la détermination des domaines dans lesquels les services écosystémiques sont en train de se raréfier, aux fins d'y orienter les efforts de préservation.

Christian Körner, de l'Université de Bâle, a parlé des changements climatiques et de la biodiversité, en se focalisant sur les espèces végétales non vivantes et sur les variations du dioxyde de carbone (CO₂). Au sujet des effets indirects du CO₂ exercés sur la biodiversité à travers les changements climatiques, il a présenté les découvertes de la recherche sur les changements climatiques survenu dans les climats froids et les écosystèmes montagneux, soulignant que les décalages microclimatiques peuvent être cinq fois plus importants que dans les scénarios émis par le Groupe Intergouvernemental sur l'Evolution Climatique. Körner a précisé que les effets du réchauffement de la surface de la planète englobent: des changements nets dans la fréquence des espèces; le fait que le climat devient plus sec dans certaines parties du globe; l'érosion des taxons à travers les sécheresses; et l'érosion des forêts due à l'augmentation du nombre des feux forestiers. Au sujet des effets directs du CO₂ sur la biodiversité, il a décrit les réponses apportées à des taxons particuliers de flore et de faune. Il a parlé du changement de focalisation opéré dans le domaine de la recherche axée sur le climat, où on est passé de la fixation du carbone aux effets exercés sur la biodiversité.

Le Président de la Table Ronde, Harold Mooney, de l'Université de Stanford et de l'Evaluation Ecosystémique du Millénaire, a affirmé qu'une nouvelle approche est nécessaire pour l'évaluation des avantages potentiels des écosystèmes, et a suggéré d'utiliser les échecs subies dans la préservation de la biodiversité pour inciter les scientifiques plutôt que le public à entrer en action.

Madeleine Tchuente, Ministre de la Recherche de Scientifique et de l'Education, du Cameroun, a parlé de la riche diversité des écosystèmes du Cameroun, et a mis en relief les initiatives entreprises dans le domaine de la préservation, et les partenariats mis en place dans le Bassin du Congo.

Au sujet de la mondialisation et de la biodiversité, Perrings a déclaré que le problème principal est celui de l'incapacité des marchés internationaux à traiter l'érosion de la biodiversité, et a suggéré d'intégrer les avis scientifiques concernant les effets environnementaux potentiellement défavorables de la libéralisation du commerce, dans les travaux de l'Organisation Mondiale du Commerce.

Laurent Piermont, Président de la Société Forestière Française de la Caisse des Dépôts, a élaboré sur les expériences vécues dans le domaine du financement de la biodiversité à travers les mécanismes de marché. Il a souligné l'importance du génie écologique à coût modéré et de la rationalisation des objectifs de la préservation.

Pierre Jacquet, Directeur Exécutif du Groupe des Agences Française de Développement, a déclaré que l'évolution des comportements sociaux et politiques est cruciale pour la préservation de la biodiversité, soulignant la nécessité de concilier les intérêts des divers acteurs et de mener davantage d'analyses économiques et sociales en rapport avec la biodiversité.

Pierre Valette, du Directeur Général de la Recherche, à la CE, a souligné la nécessité d'études sur les conséquences économiques et sociales de l'érosion de la biodiversité, assorties d'un accent sur la recherche qui aiderait à établir des seuils de durabilité.

Au cours du débat qui a suivi, abordant la question de l'utilisation traditionnelle des herbes médicinales au Cameroun, Tchuente a précisé que des instituts sont en train de mener des recherches sur les plantes utilisées à des fins sanitaires. Au sujet des coûts et des avantages de la biodiversité, Perrings a indiqué que les avantages pour le public ne doivent pas être compromis par les gains privés. Un participant a appelé les scientifiques du domaine de la biodiversité à atteindre le grand public à travers des campagnes de sensibilisation plus vigoureuses et de meilleurs programmes de communication.

Le Centre sur la biodiversité de Montréal

- [Nous contacter](#)
- [English](#)

- [Centre](#)
 - [Objectifs](#)
 - [Financement](#)
 - [Bâtiment](#)
- [Collections montréalaises](#)
 - [Herbier Marie-Victorin](#)
 - [Ouellet-Robert \(entomologie\)](#)
 - [Insectarium](#)
 - [Cercle des mycologues de Montréal](#)
- [Recherche](#)
 - [Chercheurs du Centre](#)
- [Consortium](#)
 - [Institutions partenaires](#)
 - [Chercheurs du Consortium](#)
 - [Collections participantes](#)
 - [Autres sites et bases de données](#)
- [Sensibilisation du public](#)
 - [Biodiversité menacée](#)
 - [Espace public](#)

[Accueil](#) > [Sensibilisation du public](#) > **Biodiversité menacée**

Biodiversité menacée



Amanita muscaria

La perte de la diversité biologique est devenue un enjeu planétaire. Le sentiment d'urgence qui anime la communauté scientifique, les décideurs politiques et la société civile s'explique par le fait que la Terre connaît actuellement une phase d'extinction massive. En effet, le taux actuel d'extinction d'espèces vivantes est de 1 000 fois supérieur au taux moyen observé dans l'histoire de la Terre. D'autres grandes crises d'extinctions ont affecté notre planète dans le passé, mais elles étaient dues à des changements dans les conditions climatiques. La crise actuelle est attribuable principalement aux activités humaines. D'innombrables espèces sont condamnées à l'extinction à cause de leur surexploitation, de la destruction de leurs habitats, des changements climatiques, etc.

Les scientifiques du monde entier s'accordent pour prédire l'extinction de 25 à 50 % des espèces d'ici la fin du siècle si rien n'est fait pour renverser la tendance. C'est l'héritage de milliards d'années d'évolution de la flore et de la faune qui disparaît ainsi. La conservation de la biodiversité est une responsabilité commune à tous les peuples. Sa protection est également un devoir des générations actuelles envers les générations futures. À terme, c'est non seulement l'intégrité de la biosphère qui est menacée, mais notre propre survie comme espèce. La protection de biodiversité est une condition *sine qua non* du développement durable. Si nous y manquons, c'est un nombre incalculable de bienfaits de la biodiversité que nous risquons de perdre à tout jamais. La réduction du taux de perte de la biodiversité est une condition fondamentale du développement durable. À défaut d'agir, nous perdrons à jamais l'opportunité de récolter l'ensemble des bénéfices potentiels de la biodiversité pour l'humanité.

La valeur de la biodiversité se manifeste sur plusieurs plans : écologique, social, culturel, scientifique, économique, etc. On en comprend mieux l'importance si l'on considère la part significative de la pharmacopée issue de produits extraits directement de plantes ou dérivés de produits naturels. La nature constitue une immense banque de gènes qui, à l'ère de la biotechnologie, peuvent être utilisés pour maintenir ou améliorer notre qualité de vie.

Les spécialistes en amélioration des cultures vont fréquemment chercher des propriétés agronomiques importantes pour les nouvelles variétés parmi les plantes sauvages : résistance aux maladies, aux ravageurs, aux stress environnementaux. Quand la propriété souhaitée ne peut être introduite dans les nouvelles variétés, la nature fournit souvent d'autres solutions, telles que de nouveaux biopesticides, bioengrais, etc.

Préserver la biodiversité signifie donc conserver la capacité de découvrir et de mettre au point de nouveaux médicaments et produits alimentaires et industriels. Conscients de ces enjeux, de nombreux pays ont déjà adopté des mesures pour protéger leur faune et leur flore et travailler au maintien de l'intégrité de la biosphère. Des programmes d'études ont été mis sur pied afin de comprendre pourquoi des espèces disparaissent ; d'autres pour découvrir, répertorier et conserver celles qui demeurent inconnues.

Le gouvernement du Québec a adopté un plan sur la diversité biologique de 2004 à 2007 et, plus récemment, a tenu des consultations sur son nouveau plan de développement durable. On y souligne l'importance de développer, de mettre en valeur et de diffuser les connaissances sur la biodiversité afin d'atteindre les objectifs gouvernementaux. Ces activités représentent les priorités du Centre sur la Biodiversité de Montréal.

Le Canada, pour sa part, a signé les conventions internationales sur la biodiversité et élaboré une stratégie à cet égard. Le pays participe activement au Système mondial d'information sur la biodiversité (SMIB/GBIF) via le Système canadien d'information sur la biodiversité (SCIB/CBIF) mis sur pied par Agriculture et agroalimentaire Canada. Pour le moment, cette initiative ne contient les données que d'un petit nombre de collections universitaires. Par ailleurs, le Canada n'a toujours pas adopté une approche concertée et intégrée pour la conservation et l'étude de la biodiversité. Les données sur la biodiversité

restent fragmentaires et éparpillées – une bonne partie n'est ni disponible ni accessible. Un objectif majeur du Centre sur la biodiversité de Montréal est de combler ces lacunes en collaboration avec le SCIB.

Avec la mondialisation des échanges commerciaux, de nombreuses espèces sont transportées de leur continent d'origine vers d'autres. Dans certains cas, elles s'établissent et se propagent dans les nouveaux écosystèmes avec des conséquences parfois catastrophiques pour l'humain et l'environnement. L'identification de ces espèces envahissantes, la compréhension des équilibres écologiques qui les contrôlent naturellement dans leurs écosystèmes d'origine, la mesure de leurs impacts sur les écosystèmes envahis et les options envisageables pour leur contrôle dépendent étroitement de la connaissance de la biodiversité dans tous les écosystèmes concernés.

Liens connexes

- [Sensibilisation du public](#)
- [Biodiversité menacée](#)
- [Espace public](#)

- [Centre - Informations générales du Centre sur la biodiversité...](#)
- [Collections montréalaises - Différentes collections présentes au Centre...](#)
- [Recherche - Chercheurs du Centre et leurs projets...](#)
- [Consortium - Consortium des universités canadiennes sur la biodiversité ...](#)
- [Sensibilisation du public - Moyens d'apprentissage pour le public...](#)

© 2008. Toute reproduction partielle ou intégrale du contenu du site est interdite sans l'approbation écrite du Centre sur la Biodiversité.

- Aménagement et modification du littoral
- Modification des zones intertidale et infratidale
- Extraction et déplacement de minéraux et de sédiments
- Modification de terres humides et de marais salés
- Modification des eaux de mer et des bassins versants côtiers
- Modification biologique

Aménagement et modification du littoral



Les aménagements et modifications du littoral comprennent l'aménagement urbain et la construction de ports, d'installations portuaires, des moyens de lutte contre l'érosion, de quais et de brise-lames. De façon générale, cette catégorie englobe les travaux qui modifient les caractéristiques du littoral principalement en raison du développement humain. Les mesures de protection des rives contre l'érosion naturelle ont été incluses, car elles sont souvent nécessaires pour protéger des installations situées en zones vulnérables.

Les divers paliers de gouvernement ont déjà pris des mesures pour atténuer les incidences de l'aménagement et de la modification du littoral. Les programmes d'évaluation environnementale des nouveaux projets de développement, la planification de l'utilisation des terres, les régimes de permis et les règlements municipaux sont des exemples de telles mesures.

▲Haut de page

Modification des zones intertidales et infratidales

La modification des zones intertidales et infratidales englobe les effets des engins de pêche fixes, des quais, des jetées, ainsi que de certains types de dispositifs et d'installations aquicoles terrestres. Les préoccupations ont trait à la perte d'habitats clés pour les espèces de poissons indigènes, notamment les zones nécessaires à la migration, au frai et à l'alevinage. La récolte de plantes marines dans ces zones peut donner lieu à la modification et à la perte d'habitats d'autres espèces, à la réduction de la biodiversité ou à une utilisation non durable des ressources.

▲Haut de page

Extraction et déplacement de minéraux et de sédiments

Le dragage des ports et l'élimination des sédiments sont des activités courantes sur les trois littoraux canadiens qui sont cependant réglementées afin de limiter la sévérité de leurs incidences. Le PAN intervient lorsque ces activités ont lieu dans les zones intertidales ou à proximité des côtes. L'extraction de sable et de gravier n'est pas encore courante, surtout à cause du peu de dépôts rentables. Mais cette activité peut donner lieu à une destruction localisée de l'habitat. L'extraction minière côtière n'est pas généralisée au Canada. Il est cependant arrivé que des mines et des usines de traitement situées dans les terres aient rejeté des résidus miniers et d'autres déchets dans le milieu marin; et que cela ait été source de perturbations importantes attribuables au recouvrement d'habitats voisins et à l'altération d'habitats situés en aval.

▲Haut de page

Modification de terres humides et de marais salés



Des terres humides et des marais sont modifiés et détruits depuis l'établissement en zone côtière des Européens au Canada. Il reste aujourd'hui moins de 30 % des habitats originels. Les pertes sont plus importantes dans les zones de forte urbanisation et d'agriculture intensive. Les perturbations se poursuivent sous d'autres formes, comme la récolte du foin des marais et d'autres plantes, le drainage pour lutter contre les insectes et les aménagements visant à créer des habitats pour la sauvagine.

Le Canada prend des mesures visant à protéger ces zones. Ainsi, la *Politique fédérale sur la conservation des terres humides* présente sept stratégies qui offrent une orientation pratique, de l'aide et des outils pour l'utilisation et la gestion des terres humides afin qu'elles puissent continuer à jouer leurs rôles très diversifiés (qualité et rétention de l'eau, habitats de la faune) de façon durable. La Politique traite aussi de l'étude scientifique de ces habitats et de la sensibilisation de la population.

▲Haut de page

Modification des eaux de mer et des bassins versants côtiers

Ce type de modification comprend la construction de barrages et de dispositifs de prélèvement d'eau, la pollution thermique ainsi que la dérivation des cours d'eau et l'extraction de l'eau. Cette altération des eaux marines et des bassins versants côtiers peut modifier la salinité, la température et, parfois, la quantité d'éléments nutritifs et de sédiments présents. Ces changements peuvent nuire directement à la vie marine indigène et modifier les conditions dans les habitats, notamment dans ceux qui sont les plus sensibles, comme les zones d'alevinage et les herbiers.

Au cours du siècle dernier, l'étalement urbain et le développement industriel, agricole et forestier ont exercé des pressions croissantes sur les milieux côtiers et marins et les écosystèmes avoisinants. Ainsi, la population du bassin du Bas-Fraser de la Colombie-Britannique est celle qui croît le plus rapidement au Canada.

Le Plan de gestion de l'estuaire du Fraser permet de contrer ces pressions en établissant un lien entre les organismes chargés d'adopter et de faire appliquer les mesures législatives et les politiques environnementales et ceux chargés de la gestion des terres et de l'eau. Le Plan comporte sept programmes d'action qui font l'objet d'une mise à jour permanente : l'intégration et le caractère durable, la qualité de l'eau et des sédiments, l'habitat du poisson et de la faune, le dragage et la navigation, la gestion des billes de bois, le développement urbain et industriel et les loisirs.

Au plan régional, la Commission mixte internationale réalise une évaluation de l'écoulement des eaux du lac Ontario vers le fleuve Saint-Laurent. Une partie de cette étude porte sur l'évaluation des incidences de la variation du niveau de l'eau sur certains facteurs environnementaux, comme l'érosion des berges et les dommages causés par les inondations.

▲Haut de page

Modification biologique

Les modifications biologiques des eaux côtières découlent souvent du transport accidentel d'espèces exotiques, ou non indigènes, de plancton, de crustacés, d'organismes pathogènes et de parasites présents dans l'eau des cales ou des citernes de ballast de gros pétroliers ou cargos océaniques. Ce problème résulte, en grande partie, de la vidange au large de l'eau de ballast et, de façon stricte,

étrangères découlant du rejet d'eau de ballast dans les ports. Un exemple d'une telle perturbation, même s'il n'y a pas eu d'effets directs sur le milieu marin, est la propagation très étendue de la moule zébrée dans le bassin des Grands Lacs qui a eu d'importantes incidences environnementales et économiques sur l'écosystème aquatique.

Il reste aujourd'hui moins de 30 % des habitats originels des terres humides.

La contamination d'espèces locales a provoqué la fermeture de certaines zones coquillères de la côte est du Canada. On croit que ces maladies ont été apportées par des navires, notamment avec l'eau de ballast ou les salissures se trouvant sur leurs coques, mais les sources exactes n'ont pas été démontrées.

L'expansion de la pisciculture à proximité des côtes a fait craindre la perte d'habitats par l'accumulation de déchets sous les cages, ce qui altère l'habitat à cet endroit. L'industrie piscicole est gérée par les gouvernements fédéral, provinciaux et territoriaux par l'entremise de diverses mesures législatives complémentaires.

[▲ Haut de page](#)

Résumé-Synthèse

Alain Tremblay, Louis Varfalvy, Charlotte Roehm et Michelle Garneau

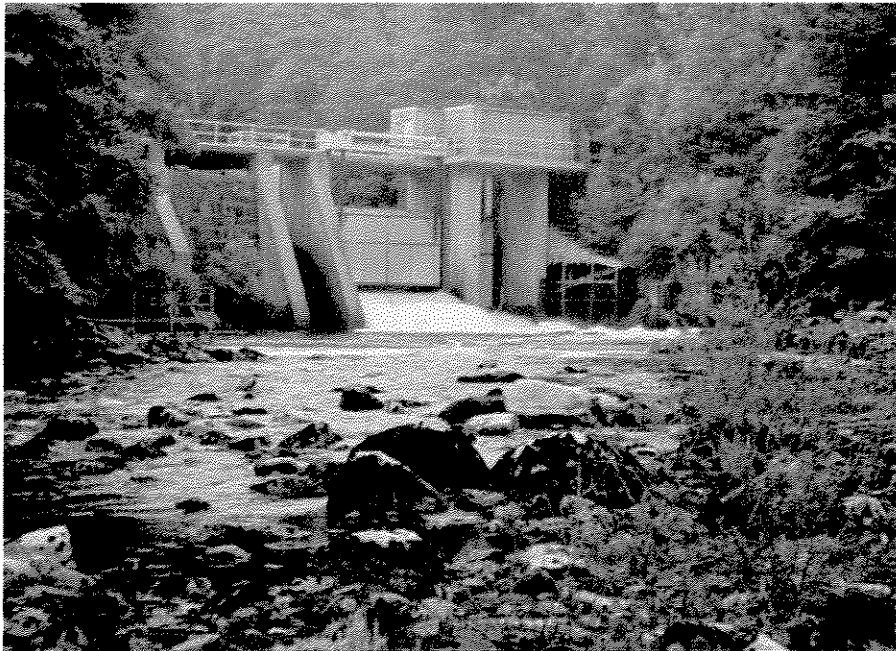
Ce chapitre a pour but de faire le point sur l'état des connaissances et d'identifier les lacunes relatives à la problématique de l'émission de gaz à effet de serre (GES) par les réservoirs hydroélectriques et les écosystèmes naturels. Il est devenu essentiel d'intégrer nos connaissances du cycle du carbone à des échelles temporelles et spatiales plus vastes de façon à mieux définir l'ampleur des flux de GES associés aux réservoirs¹ et aux écosystèmes naturels. Les données disponibles proviennent d'études à petite échelle et de courte durée (1 à 10 ans), effectuées surtout en région boréale, mais aussi en régions semi-aride et tropicale. La variabilité naturelle des flux de GES due à des variations climatiques régionales et leurs impacts sur la production biologique globale est plus importante que celle des méthodes de mesures. Il faut donc garder à l'esprit que les incertitudes concernant les flux de GES sont avant tout le résultat de variations spatiales et temporelles naturelles des flux, et non pas des techniques de mesure disponibles. La présente synthèse se base sur les résultats de plus de dix ans de suivis obtenus par différentes équipes de recherche de plusieurs universités, institutions gouvernementales et compagnies d'électricité.

¹ Pour évaluer l'ampleur des flux de GES, nous avons calculé les émissions brutes et les émissions nettes. Les émissions brutes sont celles mesurées à l'interface eau-air. Les émissions nettes des réservoirs correspondent à la différence entre les émissions brutes et les émissions naturelles des écosystèmes terrestres et aquatiques avant la mise en eau, pour l'ensemble du bassin versant, incluant la portion aval et l'estuaire. Ces définitions sont celles de WCD (2000).

Coalition Clean Baltic

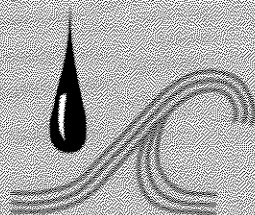
FOR PROTECTION OF THE BALTIC SEA ENVIRONMENT

The biological cost of hydropower

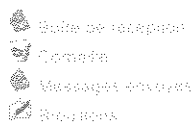


Roland Jansson
Landscape Ecology Group
Department of Ecology and Environmental science
Umeå University
SE-901 87 Umeå, Sweden

CCB Report No. 2002:2



Coalition Clean Baltic, CCB
Ostra Agatan 53, SE-753 22 Uppsala, Sweden
Tel: +46 18 71 11 55/+46 18 71 11 70
fax: +46 18 71 11 75
E-mail: secretariat@ccb.se
gunnar.noren@ccb.se
Org. number: 802015-1281, www.ccb.se



Parc national dans le Grand Nord: déjà, c'est la controverse!

Martin Croteau
La Presse

En avril, Québec a publié un plan directeur qui propose les limites du nouveau parc. Il s'étendrait sur 15 549 km² - plus de 30 fois la superficie de l'île de Montréal. Il comprendrait le lac Guillaume-Delisle, une vaste étendue d'eau qui communique avec la baie d'Hudson et dans laquelle pénètrent phoques et bélugas. À l'est, on trouverait le lac à l'Eau-claire, composé de deux bassins circulaires créés par un double impact météorique.

Ce secteur du Grand Nord est célèbre pour ses falaises de roches sédimentaires - appelées cuestas - qui atteignent des hauteurs de 400 m. Si la proposition est adoptée telle quelle, ce parc national sera le plus grand de la province.

Mais c'est ce que le plan ne contient pas qui sème la controverse. Juste au nord du futur parc, on trouve une autre splendeur naturelle, la rivière Nastapoka. Ce cours d'eau parsemé de chutes et de rapides prend sa source au lac des Loups-marins, l'un des rares plans d'eau douce au monde où l'on peut apercevoir des phoques.

«La proposition originale, qui émanait du ministère de l'Environnement et des locaux, en 2006, c'était que le parc inclue la Nastapoka, affirme John O'Driscoll, président de la Société pour la nature et les parcs du Canada. Quand les gens d'Hydro-Québec ont eu vent de cela, ils s'y sont opposés.»

Dans un mémoire présenté le mois dernier à l'occasion d'une consultation publique, la société d'État ne cache pas son intérêt pour l'exploitation hydroélectrique de la rivière. Hydro-Québec affirme avoir eu de nombreux échanges avec les ministères de l'Environnement et des Ressources naturelles dans le but d'exclure le secteur nord du parc et de permettre l'éventuel aménagement de la rivière Nastapoka.

Hydro affirme que sa position est «préventive» et qu'elle souhaite «préserver, pour les générations futures, la possibilité de développer le potentiel hydroélectrique du Nord québécois». Elle estime que la construction de barrages sur la Nastapoka pourrait générer 1000 mégawatts.

Si le chantier est mis en branle, la société d'État prévient qu'elle devra construire des routes d'accès et des pylônes électriques dans le parc national.

Contestation

La nouvelle carte du parc, qui exclut la rivière, a été vivement contestée lors des récentes audiences publiques. Le maire d'Umiujaq, une petite communauté de 415 habitants située en bordure du futur parc, fait partie des opposants. «La Nastapoka, c'est notre terrain de chasse, résume Robbie Tookalook, joint à son domicile hier. Il y a des bélugas, beaucoup de poissons, des caribous.»

Comme lui, plusieurs Inuits craignent de voir Hydro-Québec construire des barrages dans le secteur. Ils ont reçu l'appui des écologistes, mais aussi de scientifiques. Michel Allard, directeur du Centre d'études nordiques de l'Université Laval, s'insurge contre la version réduite du parc. «À notre avis, le plan directeur provisoire ne développe pas une argumentation claire et fondée sur des motifs solides de

conservation écologique pour justifier cette exclusion, écrit-il dans son mémoire. Le texte ne fait que mentionner que le parc projeté sera le plus vaste au Québec.»

http://www.lactualite.com/environnement/article.jsp?content=20080505_150657_6428&PAGE=2

Beau temps pour Hydro!

À la loterie des changements climatiques, il y aura beaucoup de perdants, mais aussi des gagnants. Hydro-Québec pourrait bien remporter le gros lot.

par Dominique Forget

Publié dans *L'actualité* du 1er juin 2008

Des précipitations accrues seraient moins rentables pour Hydro-Québec si elles tombaient uniquement sous forme de neige et s'accumulaient tout l'hiver. Déjà, au printemps, les eaux de fonte sont si abondantes que certains réservoirs n'arrivent pas à les contenir toutes. Pour protéger l'équipement, les surveillants de réseau doivent souvent ouvrir les évacuateurs de crues et laisser l'eau s'échapper en contournant les barrages. Un vrai gaspillage ! Car chaque goutte qui quitte le réservoir sans avoir fait tourner les pales des turbines représente de l'argent en moins dans les coffres de la société d'État.

Si les prévisions du MRCC pour le nord du Québec réjouissent René Roy et ses collègues, les projections pour le sud de la province sont moins encourageantes. On prévoit une diminution des précipitations au-dessus des cours d'eau, notamment des rivières des Outaouais, Gatineau et Saint-Maurice, qui alimentent chacune un chapelet de centrales. Le déficit prévu oscille entre 10 % et 15 %.

Tout compte fait, lorsqu'on additionne les avantages d'un côté et les désavantages de l'autre, le calcul indique une nette augmentation des ressources hydroélectriques pour l'ensemble de la province. Hydro-Québec devra donc repenser sa façon de gérer ses installations afin d'en tirer le maximum.

Au siège social, boulevard René-Lévesque, à Montréal, Marcel Paul Raymond dirige les activités de planification annuelle de la production de tout le réseau. De son ordinateur, il peut en quelques clics voir le niveau d'eau de chacun de ses réservoirs du territoire et de chacun des lacs et des rivières. Une fois les données saisies, il peut afficher les statistiques de production de chaque barrage et de tous les ouvrages au cours des prochains jours ou des prochaines semaines.

« Plus le niveau d'eau est élevé dans un réservoir, plus la goutte qui fait tourner la turbine est efficace. Si le niveau d'eau baisse, on ferme les vannes à un barrage, le niveau s'accroît dans le réservoir et l'eau doit être évacuée. C'est un immense gaspillage. »

La gestion de l'eau est un enjeu crucial pour Hydro-Québec. Les prévisions de précipitations sont essentielles pour anticiper les besoins en eau et optimiser la production d'énergie.

Contingent
47%
de 1242211 unités

The biological cost of hydropower

Roland Jansson
Landscape Ecology Group
Department of Ecology and Environmental science
Umeå University
SE-901 87 Umeå, Sweden

The construction of dams in rivers has brought many benefits to human societies, such as electricity, irrigation, domestic water supply, transportation and flood protection. However, an environmental cost is paid for these benefits in form of disrupted ecosystems and reduced biodiversity. Most river systems are already affected by dams, and restoring the values destroyed by dams would, if possible, be very costly and take a long time. Therefore, exploitation of rivers by new dams to produce hydropower is not an ecologically sustainable means of power production.

Most river systems are affected by dams

The majority of the world's large river systems are affected by water-flow regulation and fragmentation by dams. Seventy-seven percent of the flow in the 139 largest river systems (mean annual discharge $>350 \text{ m}^3/\text{s}$) in North America, Europe and the former Soviet Union is moderately or strongly affected by fragmentation or flow regulation (Dynesius and Nilsson 1994). Out of 13 river systems of this size in the catchment of the Baltic Sea, only one of remain unaffected by dams: The Torne-Kalix River system on the border between northern Sweden and Finland. In fact, this is the only river system of this size in Europe outside Russia that remains free-flowing.

Smaller river systems are affected by dams and flow regulation to a similar degree. In Sweden, Norway and Finland, 85% of the medium-sized river systems (mean annual discharge $40\text{-}350 \text{ m}^3/\text{s}$) are affected by dams (Dynesius and Nilsson 1994). Most of the major tributaries to regulated main river channels have dams as well. There are only two unaffected tributaries with a mean annual discharge $>100 \text{ m}^3/\text{s}$ in Sweden, Norway and Finland. Moreover, there are thousands of dams in small streams with discharges $<1 \text{ m}^3/\text{s}$. Thus, any new dams in unaffected rivers in Europe will risk destroying unique environments.

Effects of hydropower on biodiversity

The effect of dams and flow regulation on biodiversity varies strongly with the spatial scale considered. Hydropower has caused relatively few species to go extinct from an entire river system in Northern Europe. However, at the scale of river reaches, river regulation has reduced species richness, altered species composition and ecosystem functioning.

Riparian zones along free-flowing rivers harbour one of the most species-rich vegetation types in temperate to arctic biomes. The riparian vegetation is generally vertically zoned due to differences in flooding tolerance among species. A typical zonation goes from a riparian forest rich in herbs at the top, followed by a shrub zone, graminoids, and amphibious and aquatic plants at the bottom. In regulated rivers, some lakes have been dammed to act as storage reservoirs, storing water from periods of high flow to periods when the demand for

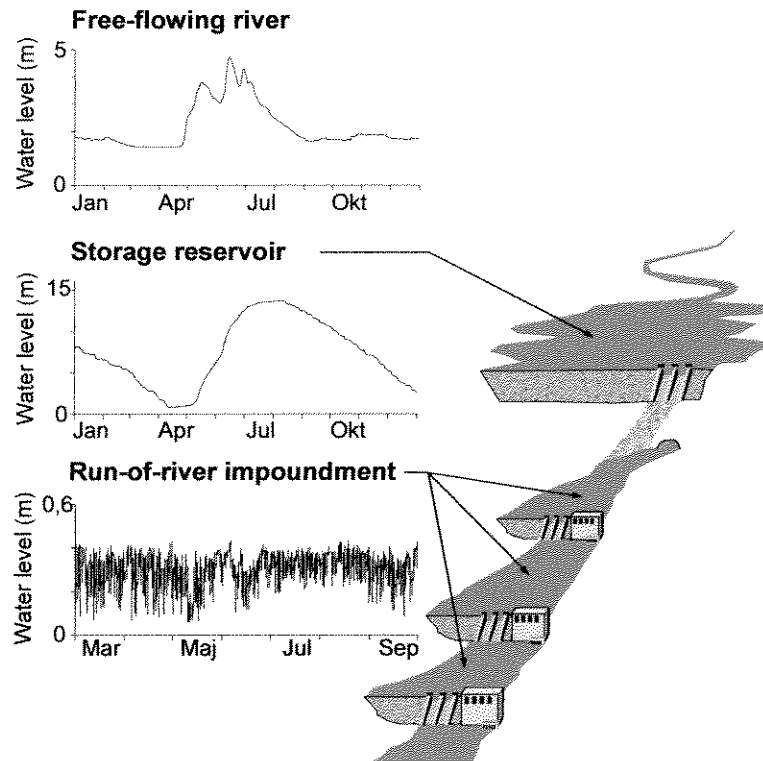


Figure 1. Schematic illustration of a regulated river and the two main types of regulated flow regimes, with free-flowing water-flow variation for comparison (top graph). In free-flowing rivers, water levels peak during spring floods, and then mostly recede until the following spring. Regulated rivers are dominated by storage reservoirs, where water is stored to be used in the following autumn and winter, and run-of-river impoundments characterised by frequent changes with small amplitude.

electricity is high (Figure 1). Water-level variations in storage reservoirs are often very large. The riparian zones are more or less devoid of vegetation, since they become successively flooded during the growing season as reservoirs are filled. Vegetation is limited to a narrow zone along the high-water level. In run-of-river impoundments, water levels vary according to the demand for water in the power station (Figure 1). Water levels fluctuate up and down daily and weekly within a range of about one meter, throughout the year. The river margins become steep and narrow, and fine material is successively eroded away. The riparian vegetation along run-of-river impoundments often forms a narrow belt without zonation close to the high water-level, and below this there are sparse occurrences of amphibious and aquatic species.

The species richness of riparian plants varies depending on the type of regulated water-level regime (Jansson et al. 2000a). In free-flowing rivers in northern Sweden, there are about 90 vascular plant species per 200-m-long stretch of river margin (Figure 2). Almost as many species are found along unimpounded reaches downstream of dams, whereas the number of species is somewhat lower along reaches that are mostly laid dry or which have reduced discharge. Along run-of-river impoundments, about one fourth of the species have disappeared, whereas one third of the species are lost from storage reservoirs, compared to free-flowing rivers (Figure 2).

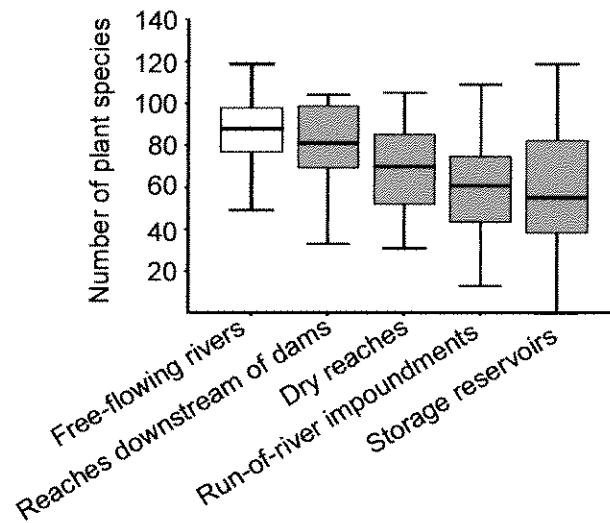


Figure 2. Box plot showing numbers of plant species per 200-m-long stretch of river margin along free-flowing rivers and water bodies with different types of regulated water-level regimes. For each type of water-level regime, 50% of the observed species-richness values are contained within the box, and 90% of the values are within the hooks. The horizontal line across the boxes mark mean values.

Flow regulation also affects macroinvertebrate faunas. The composition of bottom-living invertebrates changes and species richness decreases (Henricson and Müller 1979). In the most heavily regulated reaches (frequent short-term regulation, going down to zero discharge), both total biomass and species richness of bottom dwelling invertebrates are strongly reduced far downstream of dams. However, in less severe cases biomass may be similar to free-flowing rivers, although the faunal composition has changed. Generally, the species richness of mayflies (Ephemeroptera) and stoneflies (Plecoptera) decreases, whereas species richness of caddisflies (Trichoptera) may be comparable to free-flowing rivers, although species with net-spinning larvae are disfavoured, and only a few species remain abundant. Blackfly larvae (Simuliidae), typical at lake outlets, are generally absent from regulated parts of rivers.

Also fish faunas shift in composition in regulated rivers. In general, populations of salmonid species are reduced or destroyed, and lake-inhabiting species take over (Henricson and Müller 1979). Migration along the rivers, which is crucial in allowing fish to utilise different food resources and to reach spawning grounds, is blocked by dams. Moreover, salmonid spawning grounds are often destroyed. The reduction in fast-flowing reaches, and the increase in slow-flowing, lake-like reservoirs, leads to the replacement of salmonids by lake-inhabiting fish species, such as perch *Perca fluviatilis*, ruffe *Acerina cernua*, roach *Rutilus rutilus*, and pike *Esox lucius*, which flourish in run-of-river impoundments. Whitefish *Coregonus* spp. may be abundant in regulated rivers, but its growth is frequently impaired. Populations of grayling *Thymallus thymallus* and especially brown trout *Salmo trutta* are reduced due to competition and predation from other fishes.

Natural flow regimes create habit diversity favouring biodiversity

In a free-flowing river, different aspects of the flowing water create a mosaic of environments, providing habitats for a wide arrange of organisms (Figure 3A). The water flowing downstream carries organisms, such as seeds and aquatic insects, from upstream areas, enabling them to disperse to and colonize new areas. Organic matter and sediment are eroded from some areas, transported and deposited downstream. Fish can migrate upstream as well as downstream along the river. The river periodically floods and drains the riverbanks, due to variations in its flow. When riverbanks are flooded, they receive nutrients and sediment transported by the water, increasing the productivity of the vegetation. Some areas are cleared from litter, enhancing the development of riparian plants. The litter is redistributed into packs and piles, or end up in the river. Seeds carried by the flowing water are deposited on the riverbank. They may then germinate and establish. The floods are also a disturbance to riparian zones, eroding riparian soils, and tearing way entire plants or plant parts. This stops dominant plant species from excluding competitively inferior ones, and creates open patches for plant colonization. The result is species-rich plant communities. Organic matter from riparian zones also ends up in the river, being an important input to aquatic food webs.

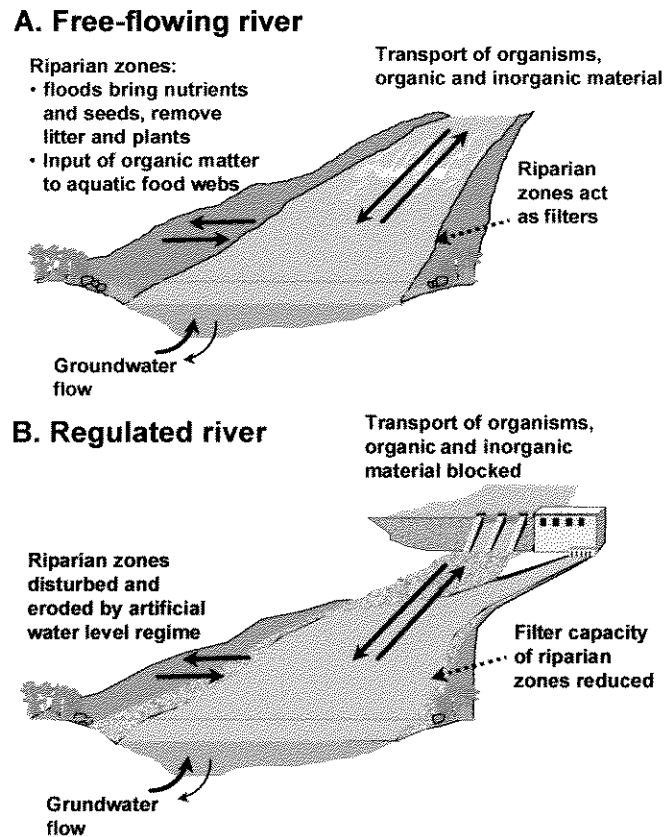


Figure 3. Examples of flows of water, matter and organisms in a free-flowing and a regulated river. Dams fundamentally change most of these flows.

Dams are barriers to the movement of organisms and transport of matter

In regulated rivers, water flows have been fundamentally changed (Figure 3B). Dams block the downstream transport of organisms, organic matter and sediments. Sediment accumulates in the reservoirs upstream of dams, and fish migration is impeded since most dams lack functional fish ladders. The artificial water-level fluctuations constitute a stress to the riparian vegetation, which largely becomes restricted to a band along the high-water level (Jansson et al. 2000a). In run-of-river impoundments, the riverbanks are frequently flooded and drained, whereas margins along storage reservoirs are successively flooded during the growing season, retarding plant growth. The ability of the sparse riparian vegetation to stop excess nutrients and toxins to enter the river is reduced, and less organic matter from the riparian zones end up in the river. As rapids and waterfalls disappear, so do organisms specialized to that habitat. Thus, the altered patterns of water flow in regulated rivers restrain the flows of energy, matter and species.

In floodplain rivers, flow regulation disrupts the connectivity between the main river channel and its floodplain (Ward and Stanford 1995), i.e. there is less transport of organisms energy and matter between them. The floodplain is no longer flooded by the river, reducing the productivity of both the river and floodplain habitats. Riparian and wetland communities on the floodplain are replaced by terrestrial plant communities. Dams block sediment transport, leading to channel deepening below the dams.

Natural rivers can be said to be corridors through the landscape, as organisms may move along or in them, thus reaching areas otherwise out of reach. Dams are barriers to such movements. Dams not only block the migration of fish, but also water dispersal by plants. In free-flowing rivers, large amounts of seeds are dispersed by the river during floods, whereas in regulated rivers, floating seeds have difficulties in passing the dams. Both the abundance and the species richness of floating seeds are lower in regulated rivers (Andersson et al. 2000). Moreover, adjacent impoundments separated by dams develop different riparian plant floras, despite that the environmental conditions of their riverbanks are similar (Jansson et al. 2000b). It is primarily species with short-floating seeds that are present in some impoundments, but missing from adjacent ones. Probably, short-floating seeds have difficulties in dispersing across dams, either because they sink or are washed ashore. In free-flowing rivers, there are no such discontinuities in the distribution of riparian plants along the rivers, suggesting that dispersal is not limiting.

Small-scale hydropower is not an environmentally friendly alternative

Small hydropower plants, often called small-scale hydropower, are often claimed to have less negative effects on the environment compared to large hydropower schemes. Few comprehensive studies of the biological effects of small-scale hydropower have been made, but available evidence indicates that *the damage per capita energy produced caused by small power plants is as bad or may be even more severe compared to large power plants.*

The physical prerequisites are the same for small as for large hydropower schemes (Figure 4). In both cases, a dam regulating the water level to the inlet of the power station is needed. The water-level variation in the water body impounded by the dam will be determined by the needs of the power station, usually implying frequent water-level changes. Below the dam, where rapids used to be, there is often a river reach which is dry or have reduced discharge. In

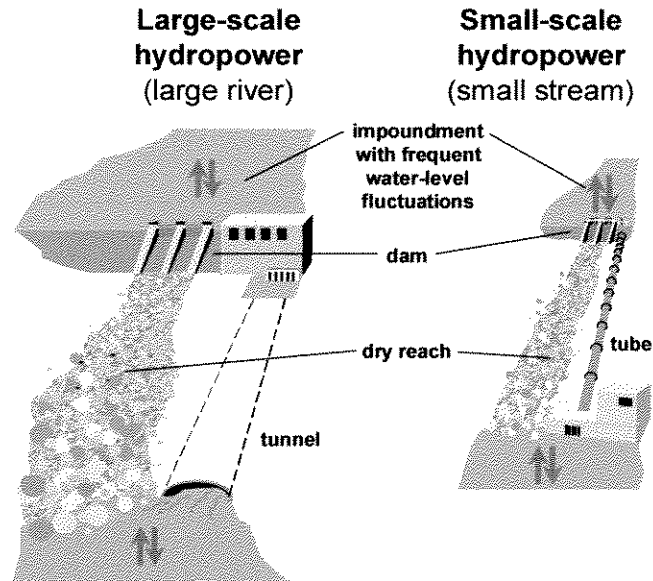


Figure 4. Comparison of large-scale and small-scale hydropower plants. Both imply similar sets of physical structures, having similar effects on river ecosystems. The main difference is the size of the affected river systems.

large rivers, water may be led by canals or tunnels from the river channel to the power station, and back again. In small rivers and streams, water is often led through tubes to power stations. In many situations, the power station is situated directly at the dam, making tunnels or tubes superfluous. The water flow in the reach below the power station is determined by operation of the power station, usually implying frequent and rapid water-level changes.

The perhaps best-documented example of the effects of small-scale hydropower on biodiversity comes from a study of the effects of changing the operation of the small, regulated stream Gunnilboån in central Sweden (Möllersten 1998). In 1996, the intensity of water-flow variation was increased, making the stream frequently vary in width from 7 to 1-2 m. In 1996 and 1997, 62 of the 198 taxa found in the stream before the new operations were commenced were missing, and the local population of stationary trout went extinct. Thus, the effects of small-scale hydropower schemes seem to be similar to large-scale ones. Small-scale hydropower schemes are small copies of large ones, applied to small rivers.

Is hydropower ecologically sustainable?

To evaluate the ecological sustainability of an energy source, it is useful to separate its environmental effects into (1) effects caused during the construction phase, (2) effects caused by the power production during the lifetime of the power plant, and (3) measures needed to restore the environment into its original or an equivalent state after power production has stopped. Hydropower by necessity means that the river and its surroundings are removed far from their original state: Lakes are dammed, land areas permanently flooded, river reaches and rapids laid dry, and canals and tunnels constructed.

Hydropower production reduces the biodiversity of river species, but is this change permanent or transient? It might be that damages to ecosystems occur primarily during the construction phase, and that ecosystems recover by time. However, the potential for recovery of riparian

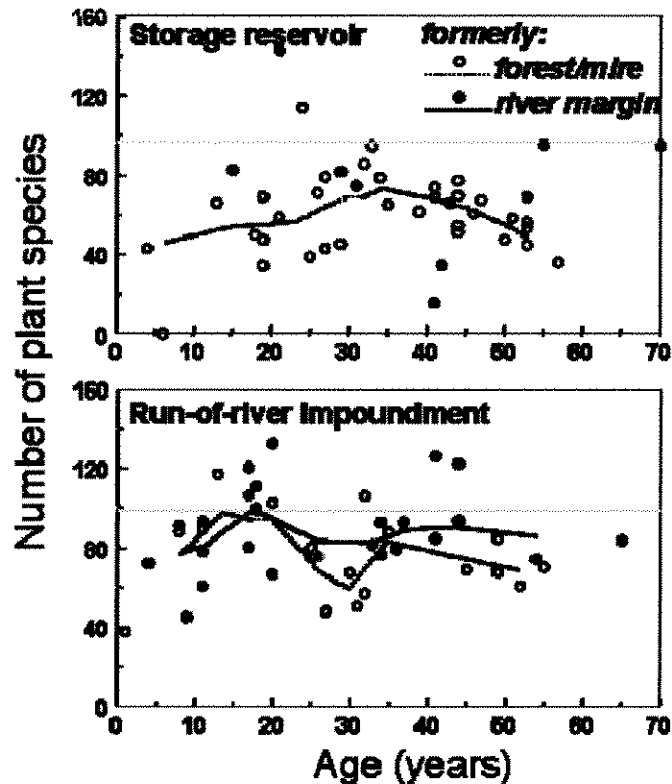


Figure 5. Numbers of plant species per 200-m-long stretch of river margin along reservoirs of different ages. Grey lines indicate mean plant species richness of comparable riparian zones in free-flowing rivers. Unfilled dots and hatched lines represent margins formed following flooding of former forests and mire, whereas filled dots and straight lines represent margins remaining in the same position as before onset of regulation.

plant communities by time is limited. The species richness of riparian plants along storage reservoirs increases the first 30-40 years after onset of regulation, but decreases thereafter (Figure 5; Nilsson et al. 1997). Plant cover drops from on average 42% of the margin to about 2%, without any signs of recovery. The reason for the rise in species richness during the first decades is that plants successively colonise the new margins. With time, fine-grained soils erode away from the upper parts of the margins, where plants may grow, and diversity is reduced.

Along run-of-river impoundments, riparian plant species richness increases the first 10-20 years, irrespective of whether the margins have developed in flooded forest/mire areas, or remain in the same position as before onset of regulation (Figure 5). After that, no clear trend is discernible. Species richness is calculated taking the available area into account, making the run-of-river impoundments appearing almost as species rich as riverbanks along free-flowing rivers. However, they encompass smaller areas than prior to regulation. Plant cover of herbs and dwarf shrubs decreases from about 50% to 30% of the margin. To conclude, the effects of regulation on species richness of riparian plants remain about 70 years after onset of regulation, and are probably permanent.

Hydropower is usually considered clean in that it does not emit any greenhouse gases or toxins. However, in cases when areas with large amounts of accumulated organic matter, such

as mires and bogs, are permanently flooded, the organic matter will eventually be released to the atmosphere in the form of methane or carbon dioxide, the two most important greenhouse gases (St. Louis et al. 2000). These emissions represent a net contribution to the atmosphere, given that few plants grow in the reservoir to accumulate the emitted carbon. The problem is considerable only in situations where the amount of organic matter dammed over is large in relation to the power produced. Organic matter often contains small amounts of mercury, which may be released as highly toxic methyl mercury when organic matter is decomposed. The mercury is accumulated in the food web, and mercury levels can be high in fish from reservoirs created by damming areas of mires and bogs (Rosenberg et al. 1995).

To restore a regulated river to something approaching the original state requires removal of the dams and restoration of the former river channel. The costs associated with doing this are huge, but despite that, there are ongoing projects to remove dams deemed to be too environmentally costly in relation to their power benefits, primarily in the USA (Bednarek, 2001). Presently, knowledge of techniques to restore rivers is insufficient to allow large-scale restoration of entire rivers. To conclude, hydropower entails strong effects on the river and its ecosystems, which will remain as long as long as power production continues, and restoration to their original states would be extremely expensive or impossible with present level of knowledge.

Conclusions – Environmental effects of river regulation

- Most river systems are already affected by dams, implying that new dams would destroy unique environments.
- The effects of dams and flow regulation on species richness vary with the spatial scale considered, and are most severe at the scale of river reaches and smaller scales.
- River regulation reduces numbers of riparian-plant species. About one third of the species per 200-m-stretch of river margin along storage reservoirs, and one fourth of the species along run-of-river impoundments are lost.
- River regulation also changes the species composition and richness of macroinvertebrates, such as aquatic insects.
- Populations of salmonid fish species are reduced or destroyed, and lake-inhabiting species such as perch and pike take over.
- Dams are barriers to the movement of organisms (e.g. fish and plants) and transport of organic matter and sediment. This impedes the formation of new habitat, which is necessary for maintaining biodiversity, and reduces ecosystem productivity.
- The damage per capita energy produced caused by small power plants is as bad or may even be more severe compared to large power plants.
- The effects of river regulation on the diversity of riverine species are not transient, but remain after 70 years and appear to be permanent.
- Although hydropower is considered not to emit any pollutants, it may under certain circumstances lead to considerable emissions of the greenhouse gases carbon dioxide and methane, and lead to accumulation of toxic methyl mercury in the food chain.
- Hydropower cannot be considered a sustainable means of power production since (1) it leads to large changes in the environment of rivers, (2) the changes remain as long as power continue to be produced, and (3) restoration would be extremely difficult and expensive.

Literature cited

- Andersson, E., Nilsson, C., and Johansson, M. E., 2000: Effects of river fragmentation on plant dispersal and riparian flora. *Regulated Rivers: Research and Management* 16, 83-89.
- Bednarek, A. T., 2001: Undamming rivers: a review of the ecological impacts of dam removal. *Environmental Management* 27, 803-814.
- Dynesius, M., and Nilsson, C., 1994: Fragmentation and flow regulation of river systems in the northern third of the world. *Science* 266, 753-762.
- Henricson, J. and Müller, K., 1979: Stream regulation in Sweden with some examples from central Europe. Pages 183-199 in Ward, J. V., and Stanford F. A., editors. *The ecology of regulated streams*. Plenum Press, New York.
- Jansson, R., Nilsson, C., Dynesius, M., and Andersson, E., 2000a: Effects of river regulation on river-margin vegetation: a comparison of eight boreal rivers. *Ecological Applications* 10, 203-224.
- Jansson, R., Nilsson, C., and Renöfält, B., 2000b: Fragmentation of riparian floras in rivers with multiple dams. *Ecology* 81, 899-903.
- Möllersten, B., 1998: Vattenkraftsutbyggnad i små vattendrag. Swedish Society for the Conservation of Nature, Stockholm.
- Nilsson, C., Ekblad, A., Gardfjell, M., and Carlberg, B., 1991: Long-term effects of river regulation on river-margin vegetation. *Journal of Applied Ecology* 28, 963-987.
- Nilsson, C., Jansson, R., and Zinko, U., 1997: Long-term responses of river-margin vegetation to water-level regulation. *Science* 276, 798-800.
- St. Louis, V. L., Kelly, C. A., Duchemin, E., Rudd, J. W. M., and Rosenberg, D. M., 2000: Reservoir surfaces as sources of greenhouse gases to the atmosphere: a global estimate. *Bioscience* 50, 766-775.
- Rosenberg, D. M., Bodaly, R. A., and Usher, P. J., 1995: Environmental and social impacts of large-scale hydroelectric development – who is listening? *Global Environmental Change* 5, 127-148.
- Ward, J. V., and Stanford, F. A., 1995: Ecological connectivity in alluvial river ecosystems and its disruption by flow regulation. *Regulated Rivers: Research and Management* 11, 105-119.

Version imprimable

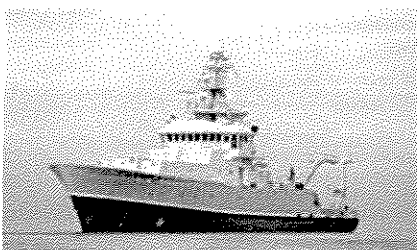
Envoyer à un ami

Contacter l'auteur

Hypoxie - Les eaux profondes du golfe du Saint-Laurent sont pauvres en oxygène dissous, tout comme celles de l'estuaire

Mario Bélanger, Université du Québec à Rimouski

le 26 août 2005



Le 26 août 2005 - Le navire Coriolis II revient d'une campagne de recherche océanographique de huit jours dans l'estuaire et le golfe du Saint-Laurent. Cet important programme de recherche mené conjointement par des chercheurs universitaires et gouvernementaux, notamment ceux de l'Institut des Sciences de la Mer à Rimouski (ISMER) et ceux de l'Institut Maurice-Lamontagne (centre de recherche québécois du Ministère des Pêches et des Océans), et soutenu en particulier par le Conseil de Recherche en Sciences Naturelles et en Génie du Canada, vise à étudier les concentrations en oxygène dissous des eaux profondes du Saint-Laurent.

On savait depuis longtemps que les eaux profondes, au-delà de 200 m dans l'estuaire et le golfe du Saint-Laurent, étaient naturellement plus pauvres en oxygène dissous que les eaux plus superficielles. Cependant, les travaux conjoints récents des chercheurs de l'ISMER et de l'IML, menés principalement dans la partie estuarienne du Saint-Laurent, ont permis de découvrir que cet appauvrissement en oxygène dissous s'était accentué au cours de la dernière décennie. Les concentrations d'oxygène dissous dans ces eaux profondes atteignaient même des niveaux que l'on peut qualifier d'hypoxique, c'est-à-dire des niveaux si bas qu'ils peuvent affecter significativement la vie et le fonctionnement des organismes vivants.

Un résultat préliminaire important de la présente mission sur le Coriolis II est que contrairement aux attentes, le phénomène d'hypoxie semble s'étendre bien au-delà de l'estuaire, soit jusque dans le golfe du Saint-Laurent. Les chercheurs ont en effet observé des conditions de très faibles concentrations en oxygène dissous également dans les eaux profondes de la partie nord-ouest du golfe du Saint-Laurent ainsi que dans le détroit d'Honguedo, au sud de l'île d'Anticosti. Même s'il reste encore beaucoup de travail à effectuer sur les échantillons et les données qui ont été récoltées lors de cette mission, il est déjà acquis que les changements observés semblent affecter plusieurs groupes d'espèces, comme par exemple les mollusques bivalves qui vivent dans ces environnements profonds. Ces phénomènes d'hypoxie sont bien connus pour plusieurs systèmes estuariens peu profonds (la baie de Chesapeake ou la mer Baltique), mais n'avaient jamais été documentés pour un environnement de l'envergure du Saint-Laurent.

Le retour du Coriolis au quai de Rimouski est prévu en matinée, le samedi 27 août 2005 entre 10h30 et 11h. Le chef de mission est le professeur Bruno Zakardjian de l'Institut des sciences de la mer de Rimouski. M. Philippe Archambault est le représentant de l'Institut Maurice-Lamontagne sur cette mission.

- 30 -

Renseignements : Mario Bélanger, UQAR (418) 723-1986 poste 1426

ETE 2008 | Vol. 5, No. 3

L'hydroélectricité a-t-elle des impacts sur la vie aquatique et marine ?

Charles-Antoine Drolet, Vice-président de Nature Québec

Comme se plaisait à le répéter un ex-premier ministre du Québec, les rivières se déversent-elles dans la mer en pure perte, ou au contraire contribuent-elles à l'importance et à la diversité de la vie marine ? On sait que les débits d'eau douce qui se déversent dans l'estuaire et le golfe ont été profondément perturbés dans le passé par les nombreux aménagements hydroélectriques qu'on retrouve dans le bassin de drainage du St-Laurent, les plus spectaculaires étant Beauharnois, Cornwall et Carillon, mais aussi Betsiamites, Péribonka, Manicouagan, Outardes, Toulouste, Ste-Marguerite et d'autres. La totalité des débits du fleuve et de ses principaux tributaires, la rivière des Outaouais et le Saguenay ainsi qu'un important pourcentage des débits des rivières de la Côte-Nord ont ainsi vu la chronologie de leur déversement dans l'estuaire et le golfe profondément modifiée. Ces transformations dans les rythmes naturels d'écoulement sont-elles sans effet ou ont-elles des impacts sur la biologie des espèces commerciales du fleuve, et sur les processus marins ? Or le fleuve St-Laurent connaît une période de transformation marquée de son débit, lié à un cycle connu de 30 ans, mais aussi semble-t-il aux effets des changements climatiques sur la pluviosité, particulièrement dans la région des Grands lacs, de sorte que dans ce contexte particulier, les modifications des débits des rivières qui alimentent le fleuve et le golfe prennent une importance relative accrue.

Victimes de développements passés : les poissons du St-Laurent

Le premier indice qui nous permet de croire que les choses ne tournent pas rond dans l'écologie du fleuve Saint-Laurent suite à la transformation de ses débits nous provient de la région du lac St-Pierre où ces variations de débits ont des impacts majeurs sur les habitats aquatiques et les populations de poissons. Des conditions de fort débit au printemps assurent un contact entre le lit principal du cours d'eau et la plaine inondable et ouvrent l'accès à des zones de reproduction ou frayères, dont la présence est essentielle au maintien des populations de poissons. (1). C'est dans la région de Sorel que le maintien de débits élevés est le plus essentiel aux activités de reproduction. Le débit du Saint-Laurent peut y varier de 5000 à 20,500 m³/sec et l'effet optimal des crues printanières se situerait à 14,500 m³/sec. Ce débit n'a toutefois été observé qu'une année sur quatre entre 1960 et 1998 en raison de son contrôle aux fins de la navigation et surtout, pour ce qui est de la rivière des Outaouais, aux fins de production hydroélectrique. L'écrêtage des crues printanières et surtout la réduction drastique de leur durée auraient privé les populations de brochet et de perchaude de 78 et de 84% de leur habitat de reproduction. Dans ces circonstances, les alevins n'ont pas le temps de quitter les sites d'éclosion avant la baisse des eaux et sont victimes de leur assèchement subséquent. Le gouvernement du Québec a dû intervenir récemment au lac St-Pierre pour tenter de corriger la situation de l'état précaire des stocks de perchaude.

Les rivières ne coulent pas en vain dans la mer : effets biologiques des rivières et des panaches estuariens

L'apport d'eau douce joue un rôle important sur la productivité des écosystèmes marins. Les interactions subtiles des apports d'eau douce et des mouvements de marées constituent la clef de la compréhension des phénomènes complexes de productivité des estuaires. Un changement majeur dans le patron de circulation d'un estuaire provoqué par l'endiguement de rivière peut avoir des effets importants sur la productivité primaire et secondaire du système. Les exigences de la croissance planctonique incluent la stabilité verticale de la colonne d'eau, suffisamment de lumière et la disponibilité d'éléments nutritifs. Tous ces facteurs peuvent être influencés par l'apport d'eau douce. La réponse n'est pas simple toutefois, car la production primaire peut être accrue ou

L'apport d'eau douce contribue à la stabilité de la colonne d'eau

L'addition d'eau douce à l'eau salée a un effet physique marqué : elle provoque une stratification plus hâtive et une plus grande résistance à la désintégration éventuelle de la stratification thermique par l'effet de brassage des marées ou du vent. Ceci a pour conséquence que dans les zones tempérées la floraison planctonique peut commencer plus tôt près des côtes où l'influence de l'eau douce est prépondérante. Thordardottir (2) a observé des variations interannuelles de l'initiation de la floraison planctonique printanière le long de la côte islandaise, et explique ces variations par l'interaction des écoulements d'eau douce et le régime des vents. Les profils de température et de salinité aux diverses stations ont montré que la stratification associée avec l'initiation de la floraison planctonique était attribuable à une réduction de salinité due aux apports d'eau douce, plutôt qu'au réchauffement des eaux de surface. On constate donc l'élément déclencheur que constituent les coups d'eau printanier et l'intérêt qu'il peut y avoir de maintenir ces patrons d'écoulement pour le maintien de la productivité marine.

D'après Mann et Lazier (3), deux mécanismes contrôlent le déplacement géographique de la floraison printanière aux latitudes tempérées : le mouvement sud-nord associé au réchauffement des eaux de surface soit le déplacement de l'isotherme 12°C, et la présence de rivières et de panaches estuariens dans les eaux côtières qui induisent une migration de la floraison printanière dans les eaux plus profondes à mesure que la saison progresse.

Les apports d'eau douce influent sur la productivité marine

K.F Drinkwater (4) résumait ainsi les conclusions d'un atelier tenu en 1986 sur les interactions des eaux douces et des écosystèmes marins (la traduction est de nous):

«Les communications présentées à l'atelier soulignent les effets marqués que les déversements des rivières dans les océans ont sur les processus chimiques et biologiques des eaux côtières. Les eaux douces initient d'importants mouvements de circulation, agissent sur la stabilité verticale, modifient les processus d'échanges et de mélange, et influencent la production d'éléments nutritifs et de production primaire. Les caractéristiques physiques et biologiques des eaux côtières reflètent la variabilité saisonnière et interannuelle des apports des rivières. L'influence sur les pêcheries est importante : on observe que les fluctuations interannuelles des rendements de certaines espèces de poissons varient en fonction des apports d'eau douce. Ces effets ne sont pas limités aux zones près des embouchures des rivières mais peuvent atteindre des milliers de kilomètres dans le cas des grands fleuves. ... Non seulement les projets hydroélectriques provoquent des altérations majeures au cycle naturel de décharge d'eau douce, mais ...ces changements peuvent causer des impacts significatifs et préjudiciables à de vastes zones marines adjacentes. »

Effets des rivières et des panaches estuariens sur la production secondaire : l'exemple du homard

Le homard est une espèce de grande importance économique pour de nombreuses communautés de pêcheur, surtout depuis l'effondrement des stocks de poissons de fond. Sutcliffe (5) a démontré qu'une corrélation excellente (une des meilleurs dans le métier) existait entre le débit du fleuve Saint-Laurent en avril et les captures de homard dans le golfe neuf années après, soit le temps que prennent les homards pour atteindre une taille commerciale. Il a conclu que la survie des larves de homard était meilleure lors des années de fort débit et que cette meilleure survie se reflétait dans les captures après cet intervalle de neuf ans (Figure 1).

Figure 1 : Prises de homard au Québec et débit du St-Laurent en avril, moyennes de trois ans, $r = .831$. D'après Sutcliffe, 1973.

Ce rapport lui a permis de conclure que le brassage de substances nutritives associées au panache estuarien est le

Les résultats ont été utilisés pour les 15 années subséquentes (6) pour des meilleures corrélations de ce type (Figure 2).

Figure 2 : Comparaison des prises de homard dans le golfe St-Laurent et des prévisions de prises à l'aide d'une régression utilisant RIVSUM soit le total des débits du fleuve St-Laurent de la rivière des Outaouais et du Saguenay. La ligne solide représente les prises, tandis que la ligne pointillée représente les prévisions. Les croix représentent les prises de 1980 à 1984. D'après Sinclair et al. 1986.

Dans le cas du homard du détroit de Northumberland, (5) il n'y a pas eu de corrélation établie avec les débits du St-Laurent, mais avec les débits du mois de juin de la rivière Miramichi, la principale rivière du secteur (Figure 3), une indication que les larves de homard produites en début de saison ont de meilleures chances de survie. Il faut toutefois préciser que ces corrélations n'indiquent pas nécessairement que des événements physiques et de nature biologique sont immédiatement inter reliés. Il y a sans doute des délais entre le moment où les débits des rivières surviennent et l'effet sur l'espèce.

Figure 3 : Production estimée de larves de homard dans le nord du détroit de Northumberland et les débits de juin de la rivière Miramichi. 1952-1963. D'après Sutcliffe, 1973.

Le débit et l'entraînement ne sont pas les seuls facteurs physiques qui influencent la productivité des homards. Le déversement d'eau douce conduit à une stratification plus hâtive de la colonne d'eau et à une plus grande résistance au mélange vertical, ce qui a pour effet que la couche de surface peut devenir plus chaude en été, effet qui peut facilement affecter la productivité et la survie des larves de homard. (3)

Les apports d'eau douce ont aussi des impacts sur les Morues et l'aiglefin.

Sutcliffe (5) s'est aussi penché sur les relations entre les débits du St-Laurent en mars et les prises d'aiglefin au Québec, en utilisant des données de 1932 à 1970. (figure 4). À la lumière de ces résultats qui indiquent une relation étroite entre les débits de printemps et la productivité d'espèces marines économiquement importantes, Sutcliffe souligne la sensibilité de ces espèces aux fluctuations des débits de rivières à des périodes critiques dans l'année et les conséquences que des modifications de débits peuvent avoir sur ces espèces.

Figure 4 : Prises d'aiglefin au Québec et débit du St-Laurent en mars, moyennes de 3 ans, délais de 10 ans. D'après Sutcliffe, 1973.

Platt et al (7) se sont aussi intéressés à l'aiglefin dans la région du banc de Nouvelle Écosse. Ils ont déterminé le moment de l'éclosion planctonique printanière (elle-même initiée par l'apport d'eau douce) pour la période de 1979 à 2001 au banc de Nouvelle Écosse où on peut trouver des larves et des œufs d'aiglefin. Ils ont déterminé qu'ils pouvaient expliquer 89% de la variance de la survie des larves par la variation de la chronologie de l'éclosion planctonique. Deux années, 1981 et 1999, qui avaient connu une éclosion planctonique particulièrement hâtive ont produit des effectifs exceptionnels.

Des résultats similaires ont été établis pour la mer du Nord. Entre 1960 et 1970, les stocks de gadidés (Morue, aiglefin, tacaud, etc.) ont quintuplé à la suite d'une bonne survie des larves provoquant ainsi une augmentation des stocks. L'hypothèse invoqué par Cushing (8) pour expliquer ce phénomène fait appel au principe de match/mismatch : une classe d'âge de poissons est forte si le maximum de la production de zooplancton survient au moment de l'apparition des larves qui s'en nourrissent. Un délai dans la production de zooplancton favorise la morue tandis qu'une production hâtive favorise l'aiglefin. Par ailleurs, Beaugrand (9) a apporté des données convaincantes qui établissent la dépendance du recrutement des morues dans la mer du Nord à la qualité, quantité, et à la chronologie de la floraison du zooplancton. Comme on l'a vu plus haut, l'apport d'eau douce peut avoir une grande importance dans l'initiation de cette floraison printanière.

poissons commerciaux se retrouvent le long des côtes sud et sud-ouest, les zones les plus affectées par l'eau douce. Thordardottir (2) a établi que les premiers stades des copépodes constituent une composante importante de la diète de la morue, et a déterminé que les années où les densités de zooplancton étaient les plus grandes étaient les années où la production de phytoplancton commençait tôt dans les conditions favorables de la stratification induite par l'écoulement d'eau douce.

Conclusions

Les effets biologiques des déversements d'eau douce peuvent être considérés sous trois aspects :

- les effets directs des matériaux transportés par les rivières sur la production biologique dans le panache ;
- l'entraînement et le brassage subséquent des eaux riches en éléments nutritifs qui vont en toute probabilité contribuer à accroître la production primaire et secondaire ;
- la stabilisation de la colonne d'eau qui a pour conséquence d'accroître la productivité lors de la floraison planctonique printanière, mais qui peut inhiber le mélange vertical et de là réduire la productivité à d'autre temps de l'année. (3) Cette floraison peut être bénéfique aux organismes qui s'alimentent dans le plancton à cette période. Les exemples du homard et de la morue illustrent cette interaction.

Les déversements d'eau douce ont donc une forte influence sur la production des estuaires et la modification de la magnitude ou du patron de ces déversements peut avoir des effets marqués sur la production biologique en milieu marin : on se prive du coup d'eau printanier bénéfique sur la productivité, et on subit les accroissements de débit en d'autre temps de l'année qui inhibent la productivité. Il faut donc conclure que l'eau des rivières ne se déverse pas dans la mer en pure perte mais qu'au contraire, elle participe ainsi aux grands cycles de la productivité marine. Les perturbations qu'ont connues les ressources marines du golfe pourraient trouver là une partie de leur origine.

Références

[1] Brodeur, P. Mingelbier, M. et Morin, 2006. Impact de la régularisation du débit des Grands lacs sur l'habitat de reproduction des poissons dans la plaine inondable du fleuve Saint-Laurent. *Naturaliste Canadien*, 130(1), 60-68.

[2] Thordardottir, T. 1986. Timing and duration of spring blooming south and southwest of Iceland. In *The Role of Freshwater Outflow in Coastal Marine Ecosystems*. (ed. S. Skreslet), pp. 345-360. Springer-Verlag, Berlin.

[3] Mann, K.H., J.R.N. Lazier. 2006. *Dynamics of Marine Ecosystems. Biological-Physical Interactions in the Oceans*. Department of Fisheries and Oceans, Bedford Institute of Oceanography. Dartmouth, Canada. Blackwell Publishing.

[4] Drinkwater, K.F. 1986. On the role of freshwater outflow on coastal marine ecosystems--- a workshop summary. In: Sigr Skreslet. *The role of freshwater outflow in coastal marine ecosystems*. Springer-Verlag Berlin

[5] Sutcliffe, W.H. 1973. Correlations between seasonal river discharge and local landings of the American lobster (*Homarus americanus*) and Atlantic haddock (*Merluccius merluccius*) in the Gulf of St. Lawrence. *Journal of the Fisheries Research Board of Canada*, 30, 353-359.

[6] Drinkwater, K.F. 1987. Sutcliffe revisited : previously published correlations between fish stocks and

[7] Blum, T., Sulyok, S., Bensch, A.M., Bensch, D.S., Müller, G. 2003. Spring algal bloom and

[8] Cushing, D.H. 1982. *Climate and Fisheries*. Academic Press London.

[9] Beaugrand, G., Brander, K.M., Lindley, J.A., Souissi, S. et Reid, P.C. 2003. Plankton effect on cod recruitment in the North Sea. *Nature* 426, 661-664.



Nature Québec

UQCN | sensible à tous les milieux

Monitoring greenhouse gas emissions from hydroelectric reservoirs in northern Quebec, Canada

Surface waters of natural freshwater bodies are capable of releasing far more carbon into the atmosphere than they absorb (Cole et al. 1994). Artificial reservoirs are also known to have environmental impacts which include greenhouse gas (GHG) emissions (Rosenberg et al. 1997, Fearnside et al. 2004, Tremblay et al. 2005). Therefore the inclusion of hydroelectric facilities as part of an overall strategy to produce clean energy requires quantification of GHGs associated with their construction. Vegetation submerged by impoundment of reservoirs ceases to function as a sink for atmospheric CO₂ and undergoes microbial decomposition, releasing both CO₂ and methane (CH₄). Reservoirs above flooded peatlands in boreal regions may release more GHGs through decomposition than reservoirs created where upland boreal forests once stood (St. Louis et al. 2000), but other factors such as climate, reservoir age, and roughness of surface waters influence emissions as well.

A Canadian research team from Environnement Illimité, Inc., Université du Québec à Montréal (UQAM), McGill University, and Hydro-Québec are attempting to answer these and other important questions in northern Québec Province as part of the Eastmain Reservoir (EM-1) Project (Fig. 1). The reservoir under study encompasses 603 km², 14%

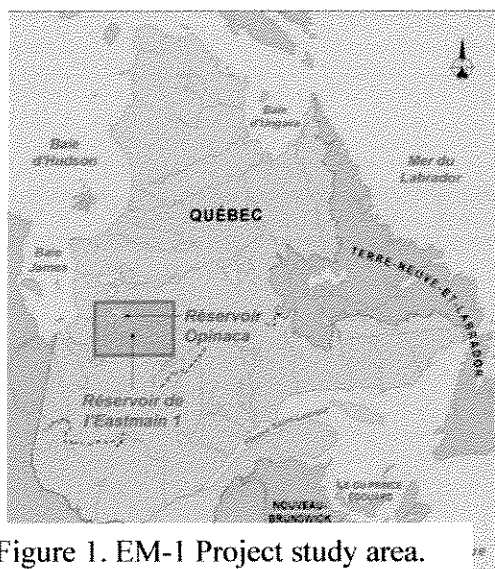


Figure 1. EM-1 Project study area.

of which is now submerged peatland. Due to the difficulty of obtaining accurate measurements in aquatic environments, researchers are relying on two separate methods to compare results (Duchemin et al. 1999).

One method employed floating static chambers (Fig. 2) that are sampled directly using the CIRAS-SC (Fig. 3) to determine changes in CO₂ concentrations at the air-water interface, monitored over 7 minute periods. The slope of the concentration over time represents the GHG flux. "Direct" fluxes are compared to theoretical CO₂ fluxes between the surface water and atmosphere calculated by the thin boundary layer (TBL) method, for which the researchers used the EGM-4 (Fig. 4) to quantify the partial pressure of dissolved CO₂ ($p\text{CO}_2$). Essentially, TBL CO₂ flux is the difference between CO₂ concentration of water and the atmosphere, as influenced by air and water temperatures and wind speed over the water's surface.

Results of the study show that CO₂ flux to the atmosphere increased significantly for the first year following impoundment of Eastmain-1 Reservoir (Bastien et al. 2007). By the second year the effect was still in evidence, although it had declined significantly. Methane production followed a similar although less dramatic pattern. Early indications are that as

reservoirs age their annual GHG emissions return to levels similar to those of natural aquatic systems (Fig. 5).

New and comparative methods like those described above are being used to estimate GHG emissions as part of a broad effort to understand the role of natural phenomena and human activity in the global carbon budget. The EGM-4 (using non-dispersive, infrared gas analysis coupled with microprocessor based signal processing) is an ideal instrument for applications that demand specificity to and a high standard of accuracy for CO₂ measurements.

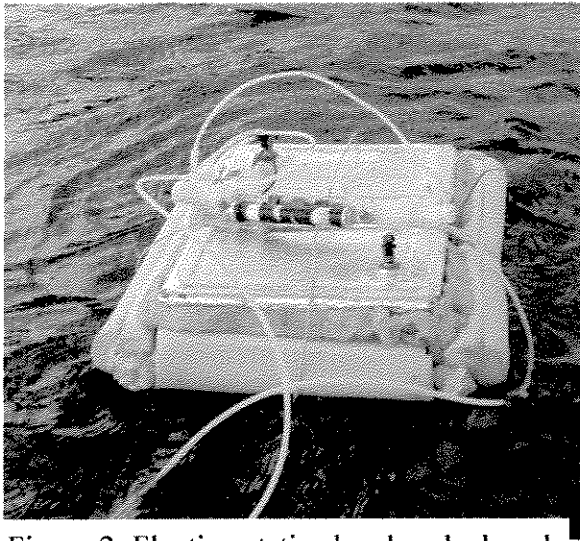


Figure 2. Floating static chamber deployed on water surface.

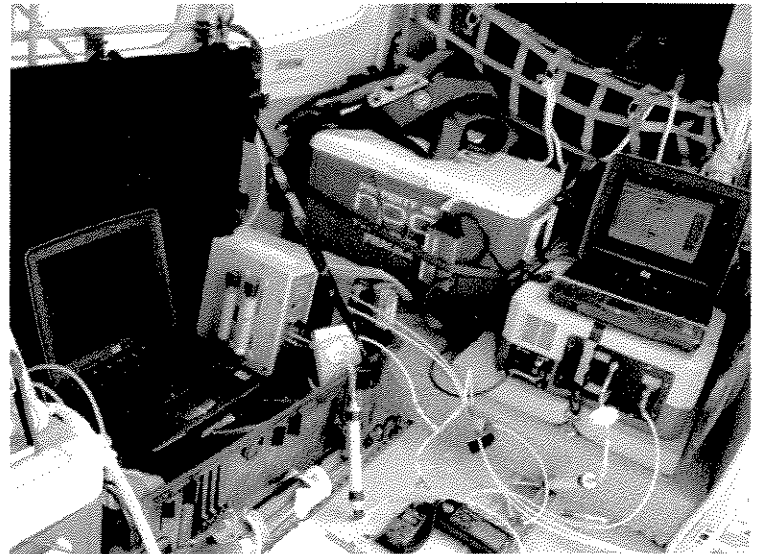


Figure 3. Suite of research instrumentation including the CIRAS-SC used to determine near-surface GHG fluxes in combination with the floating chamber.

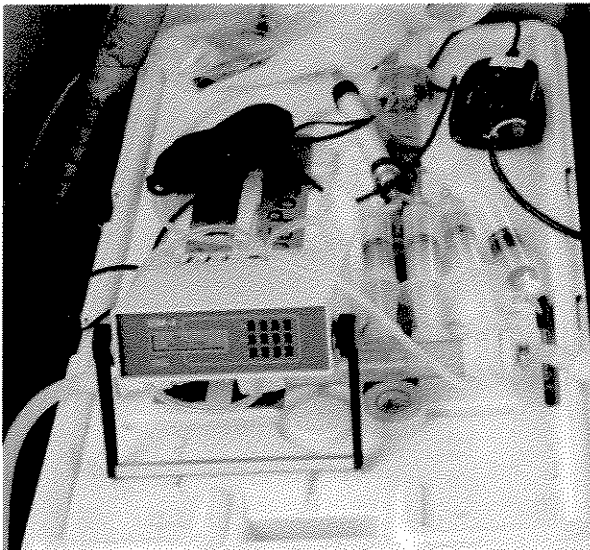


Figure 4. Thin boundary layer estimates of CO₂ flux determined from data captured by the EGM-4 gas analyzer.

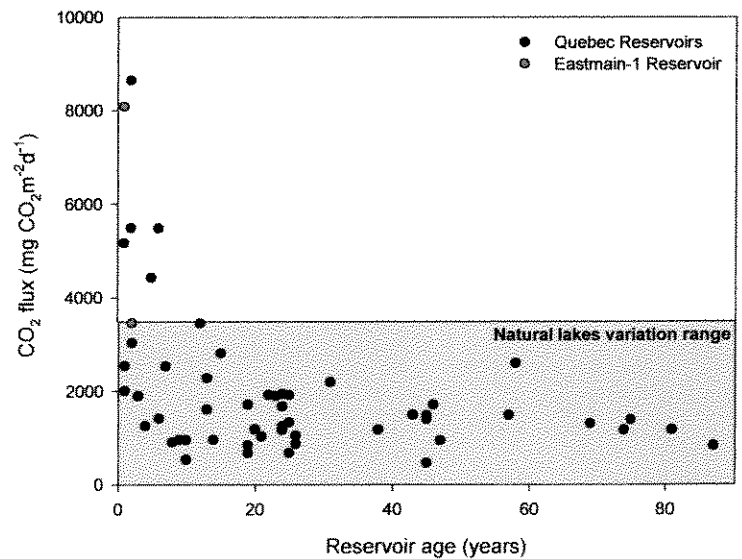


Figure 5. CO₂ flux from the newly constructed Eastmain-1 Reservoir over time as compared to flux rates of reservoirs in Québec Province.

Acknowledgements:

Thanks to Julie Bastien, Project Leader, Environnement Illimité, Inc., for background information and use of photos and data.

References:

- Bastien, J, Côté, A, Tremblay, A, 2007. Aménagement hydroélectrique de l'Eastmain-1 - Étude des flux de gaz à effet de serre - Résultats été 2007. Joint report from Environnement Illimité inc. and Hydro-Québec Production, Direction Barrages et Environnement. 49 p. and appendices.
- Cole JJ, Caraco NF, Kling GW, Kratz TK, 1994. Carbon dioxide supersaturation in the surface water of lakes. *Science* 265: 1568-1570.
- Duchemin E, Lucotte M, Canuel R, 1999. Comparison of static chamber and thin boundary layer equation methods for measuring greenhouse gas emissions from large water bodies. *Environmental Science and Technology* 33: 350-357.
- Fearnside, PM, 2004. Greenhouse Gas Emissions from Hydroelectric Dams: Controversies Provide a Springboard for Rethinking a Supposedly 'Clean' Energy Source. *Climatic Change* 66(1-2):1-8.
- Rosenberg, DM, Berkes, F, Bodaly, RA, Hecky, RE, Kelly, CA, Rudd, WJM, 1997. Large-scale impacts of hydroelectric development. *Environmental Reviews* 5(1):27-54.
- St. Louis, VL, Kelly, CA, Duchemin, E, Rudd, JWM, Rosenberg, DM, 2000. Sources of greenhouse gases to the atmosphere: A global estimate. *BioScience* 50(9):766-775.
- Tremblay, A, Therrien, J, Hamlin, B, Wichmann, E, LeDrew, LJ. 2005. GHG emissions from boreal reservoirs and natural aquatic ecosystems. In: Tremblay, A, Varfalvy, L, Roehm, C, Garneau, M, (eds). *Greenhouse Gas Emissions - Fluxes and Processes: Hydroelectric Reservoirs and Natural Environments*. Springer, pp. 209-232.

For further technical support, please contact us at:

PP Systems
110 Haverhill Road, Suite 301
Amesbury, MA 01913 U.S.A.

Tel: +1 978-834-0505

Fax: +1 978-834-0545

Email: sales@ppsystems.com

URL: www.ppsystems.com

How Dams Vary and Why It Matters for the Emerging Science of Dam Removal

N. LEROY POFF AND DAVID D. HART

Dams are structures designed by humans to capture water and modify the magnitude and timing of its movement downstream. The damming of streams and rivers has been integral to human population growth and technological innovation. Among other things, dams have reduced flood hazard and allowed humans to settle and farm productive alluvial soils on river floodplains; they have harnessed the power of moving water for commerce and industry; and they have created reservoirs to augment the supply of water during periods of drought. In the 5000 or so years that humans have been building dams, millions have been constructed globally, especially in the last 100 years (Smith 1971, WCD 2000).

If dams have successfully met so many human needs, why is there a growing call for their removal? The answers to this question require an appreciation of society's changing needs for, and concerns about, dams, including the emerging recognition that dams can impair river ecosystems (Babbitt 2002). But decisions about dam removal are complex, in no small part because great scientific uncertainty exists over the potential environmental benefits of dam removal. Certainly, the scarcity of empirical knowledge on environmental responses to dam removal contributes to this uncertainty (Hart et al. 2002). More fundamentally, however, a scientific framework is lacking for considering how the tremendous variation in dam and river attributes determines the ecological impacts of dams and the restoration potential following removal. Such an ecological classification of dams is ultimately needed to support the emerging science of dam removal.

In this article, we develop a conceptual foundation for the emerging science of dam removal by (a) reviewing the ways that dams impair river ecosystems, (b) examining criteria used to classify dams and describing how these criteria are of limited value in evaluating the environmental effects of dams, (c) quantifying patterns of variation in some environmentally relevant dam characteristics using governmental databases,

AN ECOLOGICAL CLASSIFICATION OF DAMS IS NEEDED TO CHARACTERIZE HOW THE TREMENDOUS VARIATION IN THE SIZE, OPERATIONAL MODE, AGE, AND NUMBER OF DAMS IN A RIVER BASIN INFLUENCES THE POTENTIAL FOR RESTORING REGULATED RIVERS VIA DAM REMOVAL

(d) specifying a framework that can guide the development of an ecological classification of dams, and (e) evaluating the ways that dam characteristics affect removal decisions and the future of dam removals. We restrict our analysis to the United States, where dam removals are currently hotly debated; however, the ecological framework we advocate could also be generalized to other parts of the world.

How dams impair river ecosystems

Although the rationale for dam removal often includes a range of social and economic concerns (RAW/TU 2000), the central justification for removing dams from an environmental perspective is that they adversely impact the structure and function of river ecosystems. Both individually and cumulatively, dams fundamentally transform river ecosystems

N. LeRoy Poff (e-mail: poff@lamar.colostate.edu) is an associate professor at Colorado State University, Fort Collins, CO 80523, where he teaches and conducts research in riverine ecology. David D. Hart (e-mail: hart@acnatsci.org) is an ecologist, academy vice president, and director of the Patrick Center for Environmental Research at the Academy of Natural Sciences, Philadelphia, PA 19103. © 2002 American Institute of Biological Sciences.

in several ways: (a) They alter the downstream flux of water and sediment, which modifies biogeochemical cycles as well as the structure and dynamics of aquatic and riparian habitats. (b) They change water temperatures, which influences organismal bioenergetics and vital rates. (c) And they create barriers to upstream–downstream movement of organisms and nutrients, which hinders biotic exchange. These fundamental alterations have significant ecological ramifications at a range of spatial and temporal scales.

Local effects. The local, or site-specific, alterations caused by dams, especially very large dams, have been studied extensively over the last few decades (Ward and Stanford 1979, Petts 1984, Ligon et al. 1995, Collier et al. 1996, Pringle et al. 2000). Storage of water and capture of sediment by dams cause profound downstream changes in the natural patterns of hydrologic variation and sediment transport. Numerous ecological adjustments follow. For example, reduction in the magnitude of downstream peak flows typically isolates the main channel from the floodplain, resulting in reduced recruitment of riparian species (Scott et al. 1996) and reduced access to floodplain habitats for fishes (Bayley 1995). Long-term storage and nonseasonal release of floodwaters can severely alter downstream food webs and aquatic productivity (Wootton et al. 1996). Many hydropower dams operate to produce dramatic daily flow variation that effectively reduces downstream habitat and aquatic productivity (see Poff et al. 1997 for examples). Water released from the reservoir may carve into the downstream river channel as it reestablishes its transport capacity, causing channel incision and isolating it from adjacent floodplains or tributary outlets (Petts 1984, Collier et al. 1996). Fine sediments are preferentially transported, often resulting in an excessive coarsening and armoring of the riverbed and a reduction in habitat quality for bottom-dwelling organisms.

If reservoirs exceed a certain depth and flows are slow enough, thermal stratification can occur. Deep waters can have very different temperatures than those on the surface, often maintaining temperatures near 4°C. Thus, downstream from reservoirs that release this deep water, the thermal regime is characteristically “summer cool, winter warm.” Because temperature directly affects the growth and developmental rates of aquatic organisms, such altered thermal regimes greatly modify the densities and kinds of species present. This new downstream regime is favorable for cold-adapted species like trout, and warm-adapted species often diminish in abundance or are lost (Ward and Stanford 1979). Thermal alteration and biological disruption can persist for tens of kilometers (km) downstream, depending on downstream tributary inflows (Muth et al. 2000).

Landscape effects. Dams occur so frequently in many watersheds that their cumulative ecological effects are likely to be profound, although this idea has received less attention than studies of individual dams. For example, Benke (1990) reported that there are only 42 high-quality, undammed

ivers longer than 200 km remaining in the continental United States, and Wisconsin has an average of one dam for every 14 km of river (WDNR 1995). The extensive fragmentation of free-flowing rivers promotes ecosystem isolation. The imperiled status of many salmon stocks in the Pacific Northwest is in part attributable to the gauntlet of dams these fish encounter in their migrations to and from the ocean (NRC 1996). Fragmentation also prevents the dispersal and persistence of inland species. For example, the diversity of European riparian communities is probably reduced because of the interruption by multiple dams of the downstream transport of water-dispersed seeds (Nilsson and Berggren 2000). Prevention of exchange among isolated populations may also imperil inland fish populations and other species such as mussels (Pringle et al. 2000, Fausch et al. 2002).

Water storage and sediment capture by thousands of dams has also measurably altered earth surface processes at regional and global scales (Graf 1999, Rosenberg et al. 2000). For example, the suspended-sediment loads carried by the Mississippi River to the Gulf of Mexico have decreased by one-half since the Mississippi Valley was first settled by European colonists, mostly from the construction since 1950 of large reservoirs on the sediment-laden Missouri and Arkansas rivers (Meade 1995). Other associated cumulative effects of dams that have either been demonstrated or postulated include alteration of sea level (Chao 1991), generation of greenhouse gases (St. Louis et al. 2000), and disruption of the hydrologic flux to the oceans (Sahagian 2000).

Criteria used to describe dams and their scientific limitations

Several criteria are used to characterize dams from an engineering perspective. Some of these criteria bear more strongly on the issue of dam removal and river restoration than others. Chief among these are the size of a dam, its operational purpose, and its age. Dam size not only influences such engineering considerations as construction and repair costs, it also affects the potential range and magnitude of ecological disturbances to the aquatic ecosystem (ASCE 1997). A dam's operational plan influences the type, magnitude, frequency, and timing of environmental impacts on the riverine ecosystem. The age of a dam can affect structural repair costs, as well as the cumulative magnitude of downstream channel alteration because of sediment accumulation within the impoundment. Traditionally, dam size and operational type have been discussed among engineers in simple categorical terms, such as small versus large dams, or storage versus run-of-river dams. In reality, these characteristics are more continuous and multidimensional, and it will be important to analyze and synthesize this complexity in developing an ecological classification to support the emerging science of dam removal.

Dam size. Structures have generally been small for most of the history of dam building, reflecting preindustrial techni-

cal skills and agrarian social needs. During the 19th and 20th centuries, however, new technologies allowed the construction of much larger and more complicated structures to generate hydroelectricity, control floods, provide drinking water, support large-scale irrigation, and improve navigation (Smith 1971, Schnitter 1994). In the United States, the pace of dam building accelerated dramatically after World War II, though relatively few dams have been constructed in the last 10 to 20 years (Graf 1999). It is during this period of building large dams that the burgeoning scientific understanding of the environmental impacts of river regulation has developed, with its focus on the large structures that dramatically alter riverine ecosystems. Yet most of the dams on the planet are relatively small structures, and evaluation of their environmental impacts is critical to the issue of dam removal.

Dams vary tremendously in size (height and width) and hence in their reservoir storage volume, factors that have very important direct and indirect environmental impacts (see below). Thus it is very tempting to use size as a primary descriptor of a dam's potential ecological impact. Unfortunately, the criteria used by governmental agencies and organizations to classify dam size do not adequately reflect this variation, and these criteria are not always used in a consistent manner. For example, the US Army Corps of Engineers' National Inventory of Dams (USACE 2000) emphasizes dam safety and defines dams as large if they meet one of three criteria: (1) a high hazard potential (i.e., likely loss of human life if the dam fails), regardless of the dam's absolute size; (2) a low hazard potential but height exceeding 7.6 meters (m) and storage capacity greater than 18,500 cubic meters (m³); or (3) a low hazard potential but height exceeding about 1.8 m and storage exceeding 61,700 m³. Other organizations have adopted quite different criteria for defining dam size. For example, the International Commission on Large Dams classifies dams as large if either their height exceeds 15 m or their height is between 5 and 15 m and a reservoir greater than 3×10^6 m³ is impounded (WCD 2000). Yet another classification defines hydropower dams as either low-head or high-head, depending on whether their height is less than 30 m or greater than 30 m, respectively (EnergyIdeas 2001). The criteria for classifying dams even differ among states.

There are at least two reasons why these criteria are problematic for defining dam characteristics from the perspective of environmental effects. First, as illustrated above, the same dam can be classified as large according to one definition and small according to another. Second, even if only one definition is adopted, dams that are grouped together can vary tremendously in size. For example, the USACE (2000) database of large dams includes structures with heights ranging from less than 2 m to more than 200 m, and storage volumes

from less than 100 m³ to 3.7×10^{10} m³. Such marked differences in dam size will necessarily translate into very different uses and environmental effects.

Dam operations. Although designed to meet many different human needs, the two basic functions of dams are to store water and raise water levels (McCully 1996). The storage ability of dams allows runoff to be retained for subsequent controlled release, whereas the ability to raise upstream water levels permits water diversion, increases hydraulic head for hydropower generation, creates impoundments for recreation, and so on. The most common classification of operational characteristics divides dams into two groups, storage and run-of-river, based in large part on these functional differences (USBR 2001). For example, a storage dam typically

has a large hydraulic head and storage volume, long hydraulic residence time, and control over the rate at which water is released from the impoundment. By contrast, a run-of-river dam usually has a small hydraulic head and storage volume, short residence time, and little or no control over the water-release rate (EPA 2001).

As with dam size, however, this dichotomous classification has several limitations. First, different criteria are sometimes used to place

dams in an operational class. For instance, the state of Pennsylvania defines run-of-river dams as relatively small structures whose impoundments are confined completely within the banks at normal flow levels (Pennsylvania Fish and Boat Commission 2001), a much more restricted definition than that used by most federal agencies. Second, membership in a single class can conceal large and important variation. For instance, storage dams can include flood-control dams that dramatically alter seasonal flow patterns, as well as hydropower dams that impact flow regimes primarily on a time scale of hours to days, in response to fluctuating electrical demand. Likewise, run-of-river dams can have whole-reservoir turnover times ranging from a few hours to many weeks, and impoundment depths ranging from 1 m to more than 30 m. Finally, many "multipurpose dams" are used for flood control, irrigation, navigation, power generation, and recreation and do not fit neatly in either operational class.

Despite the challenges involved in creating a simple classification system that effectively describes variation in the size and operational characteristics of dams, such variation can have markedly different ecological effects (Hart et al. 2002). For example, the flow regime below a flood-control dam 50 m high will be moderated to reduce peak flows, increase base flows, and alter natural seasonal timing of flow variations (Petts 1984). By contrast, a run-of-river hydropower dam that is 10 m high may only occasionally modify peak flows and is unlikely to substantially alter thermal regimes downstream;

Dams vary tremendously in size (height and width) and hence in their reservoir storage volume, factors that have very important direct and indirect environmental impacts....

however, it will capture the coarser fraction of transported sediment. Very small dams, such as a 2-m-high diversion dam and run-of-river mill dam, are likely to have relatively limited effects on peak flows or downstream sediment regime by virtue of their small storage volume, although they may still reduce low flows downstream and prevent upstream movement of small fishes. Thus the development of a more complete understanding of dam effects, as well as responses to dam removal, will require improvements in our ability to characterize variation in ecologically important dam characteristics such as size and operational mode.

Damage. Dams have a finite life span, so dam age can be an important factor affecting removal decisions. Two of the major factors influencing the aging process are the deterioration of construction materials and the accumulation of sediment within the dam's impoundment.

Infrastructure safety and repair. As dams age, they become more prone to failure. For example, the failure of three dams during the 1970s (Buffalo Creek, Teton, and Toccoa Creek) resulted in 175 fatalities and more than \$1 billion in losses (ASCE 2001a). More recently, heavy rains from a single tropical storm in 1994 caused more than 230 dams to fail in Georgia (FEMA 2001). Because of the boom in US dam construction that occurred from 1950 to 1980, we now face problems stemming from aging dams. This challenge is exacerbated by the fact that one-third of high-hazard dams have not even undergone safety inspections in the last 8 years (ASCE 2001b). Although the failure of a small dam may threaten fewer lives and cause less property damage than a large dam, many small dams are much older and in poorer condition than large dams. Of course, the life span of some dams can be substantially increased by continuous maintenance, but the associated costs can be high. For example, the cost of repairing a small dam can be as much as three times greater than the cost of removing it (Born et al. 1998). We emphasize, however, that the relative costs of repair and removal are likely to vary markedly, depending on the regulatory policies of different states, especially as they address potential concerns about the quantity and quality of accumulated sediments. Nevertheless, these safety and repair issues underscore the challenges of maintaining an aging dam infrastructure.

Sedimentation. Sediment capture by dams reduces reservoir storage capacity and impairs dam functionality. For modern dams, this process generally happens at a much faster rate than the loss of structural integrity of construction materials. Thus sedimentation is often a factor limiting a dam's useful life (Morris and Fan 1998). For example, high sedimentation rates have reduced the storage capacity of Matilija Dam in southern California by about 50% since it was built in 1948 (Matilija Coalition 2000). By contrast, some dams with low sedimentation rates have remained functional for extremely long periods, in some cases up to many hundreds of years (Schnitter 1994).

The importance of sedimentation is now widely recognized, but sedimentation rates were not consistently factored into

dam design criteria until the 1960s (Morris and Fan 1998), and many dams are expected to fill in with sediment at rates exceeding design expectations (Dendy 1968). Sedimentation rates vary greatly from watershed to watershed, however, because of spatial variation in sediment supply and delivery that is controlled by basin geology, slope, drainage density, and land use or cover. Erosion occurs largely in response to large precipitation events, so climate is also an important controlling factor in dam aging. Engineers now typically design reservoirs to incorporate a 100-year sediment storage pool, but human disturbance of land surfaces can greatly increase sediment yield and thus reduce a reservoir's effective life span. For example, sediment yield can increase by two orders of magnitude in regions with extensive road construction (Morris and Fan 1998).

Patterns of variation in dam characteristics

Various agencies and organizations are responsible for maintaining inventories of dams and their characteristics, particularly for purposes such as dam safety and water supply. For example, the International Commission on Large Dams has a global inventory of about 45,000 large dams (WCD 2000). In the United States, the Army Corps of Engineers maintains the National Inventory of Dams (USACE 2000), which includes more than 76,500 "large" structures. In addition to these structures are an estimated 2,000,000 or more "small" dams in the United States that are not included in this national database (Graf 1993). Information for these smaller structures is compiled and maintained largely by state regulatory agencies and is therefore much more dispersed and uneven in geographic coverage. Indeed, only a few states have compiled comprehensive state-wide electronic databases for these smaller structures.

We examined variations in characteristics of dams in the federal database and then compared them with dam characteristics for two states, Wisconsin and Utah. The size (height) distribution of federally cataloged dams is illustrated in figure 1. Almost half the dams in the federal database are in the 4 to 16 m height range. The smallest dams (< 2 m) are relatively rare in the federal database, especially when compared with their estimated abundance on the landscape (Graf 1993). Dams in different parts of the United States are often operated in a different fashion because of regional variation in climate and economic activity. Such operational differences are clearly seen by dividing the United States into eight geographic regions that reflect broad differences in physical setting (climate, topography) and settlement history (figure 2).

The picture of operational purposes of dams shown in figure 2 is unlikely to represent operations for the 2,000,000 or so smaller dams that are not in a national database. In an effort to evaluate this expectation, we analyzed data for Wisconsin and Utah, two states that have relatively complete inventories and that differ markedly in climate and topography. These two states might offer some measure of the range of variation in operational purposes of small dams (although we

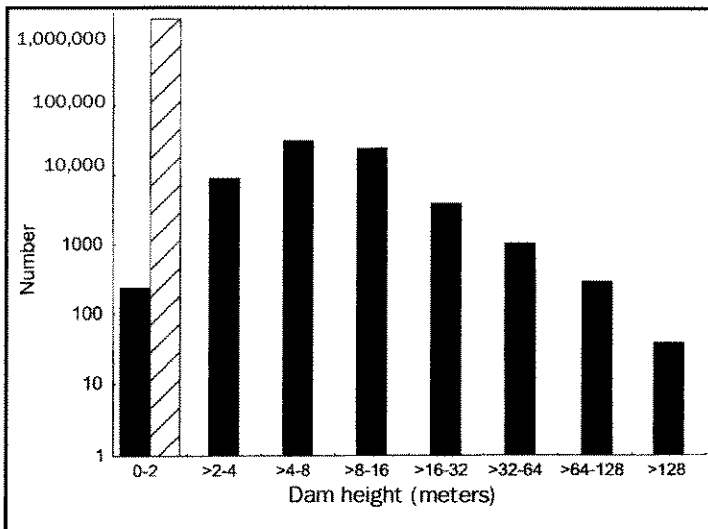


Figure 1. Distribution of US dams by structure height. Data are from the National Inventory of Dams (solid bars; USACE 2000) and estimated by USACE for dams less than 2 meters in height (diagonally hatched bar; Graf 1993).

do not argue they are statistically representative of the United States as a whole). By comparing the overlap of dams in these statewide databases with the more comprehensive national database, one can get a sense of the adequacy of using the national database to evaluate the distribution and function of the much more numerous small dams, which are more likely to be prime candidates for removal in the future.

Figure 3a compares the size distribution of the 3843 Wisconsin dams for which height is recorded in the state database with the 655 Wisconsin dams listed in the national database (USACE 2000). As expected, the national database under-represents the proportion of smaller structures (< 2 m) and overrepresents the proportion of larger structures (> 8 m). Moreover, the correspondence between the state and national databases in terms of operational purpose is poor. Most (39.4%) dams are classified by the state as “protection, stock or small farm pond,” a use category represented by only 2% in the national database. By contrast, the national inventory overestimates recreation, fish and wildlife ponds, flood control, and hydropower categories, but is reasonably representative for dams classified as irrigation, which is not a major use in Wisconsin (data not shown).

In the Utah database, 1641 dams are listed, of which only 104 are included in the national inventory. As shown in figure 3b, the size distribution of dams in the state database is very poorly represented by the national database, with the proportion of dams less than 4 m in height being under-represented and dams greater than 8 m in height being over-represented in the national database. In both the state and national databases, dams designated as primarily irrigation are the most prevalent use category (data not shown), although the national database overestimates their proportional representation by a factor of two relative to the state database. Stock ponds constitute 22% of state-identified dams, but are

completely absent from the national inventory. Similarly, the national database underestimates the occurrence of flood control structures in Utah by a factor of six relative to the state database.

Thus, in summary, the national database for large dams does a relatively poor job of characterizing small dams in terms of size distribution and operational purpose for both Utah and Wisconsin.

The need for an ecological classification of dams

A formal characterization of how dams modify river ecosystems represents a major scientific challenge, especially because the type and magnitude of environmental alteration stems from interactions among natural processes, dam characteristics, and management practices. At present, little empirical data are available to allow meaningful generalization. This reflects, in part, the fact that readily available, simple descriptors for dams (e.g.,

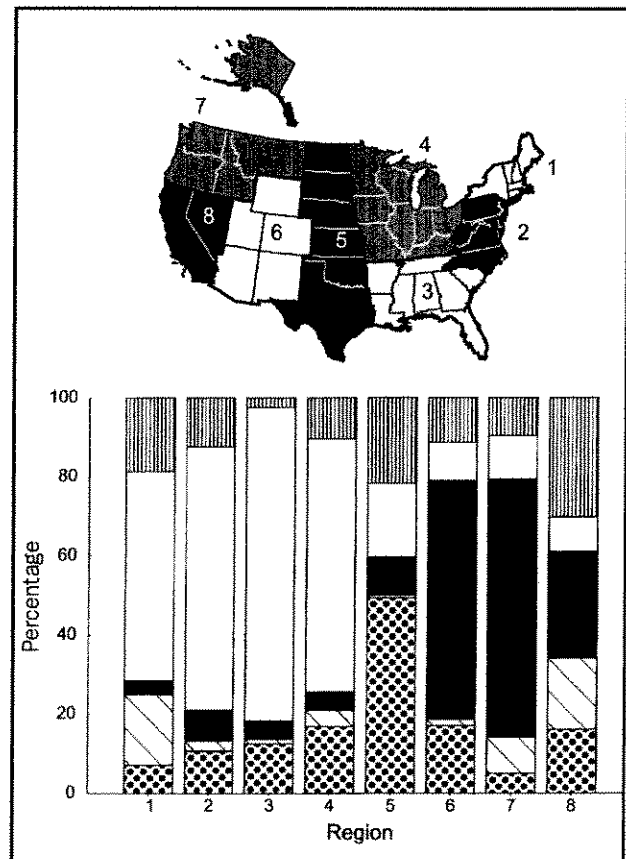


Figure 2. Percentage distribution by geographic region of dams falling into five categories of primary operational purpose, as defined in the national inventory of dams (USACE 2000). Dam uses are defined as flood control (stippled), hydropower (diagonally hatched), irrigation (solid black), recreation (solid white), and public supply (vertically hatched). These five uses represent 71% of the dams (54,903 dams).

2000) have reliable reported values for reservoir storage volume.

Indirect measures of HRT might provide an avenue for dam characterization; however, such measures are themselves limited. For example, in natural lakes, about 33% of the variation in HRT is statistically explained by variation in lake volume (Kalff 2002), so an indirect measure of reservoir volume might provide a rough estimate of HRT. Unfortunately, the most reasonable predictor variable, dam height, is only weakly correlated ($r^2 = 0.21$ for log-log data) for that portion of the national database containing values for both variables. Thus HRT is unlikely to be predicted meaningfully from dam height. In natural lakes, the unexplained 67% of the variation between HRT and lake volume probably reflects differences in regional runoff patterns and in lake morphometry (surface area to volume ratio) (Kalff 2002). Similarly, with reservoirs, regional differences in inflows will affect HRT. For example, Graf (1999) estimated maximum reservoir capacity (m^3) to store mean annual runoff (m^3 per year) to range from 0.25 to 0.37 years of storage in the upper Midwest and Northeast to 3.8 years in the arid Southwest. These values provide a sense of how HRT is regionally variable; however, predicting HRT for individual reservoirs will require that operational mode also be taken into account, since human control over dam outflows are a determinant of active reservoir storage and HRT.

Ultimately, efforts to categorize dam operations (and thus key variables like HRT) from a scientific perspective must account for differences in management practices that reflect variable social settings, economic conditions, and human preferences. Beyond the regional differences in climate and runoff, individual reservoirs are often managed for multiple purposes that can vary over time. Clearly, different types of operations can have very different environmental effects. For example, flood storage dams are often drawn down before a predictable flooding season and they are thus able to store peak flows, thereby modifying downstream flow and sediment regimes. Run-of-river dams of similar size, by contrast, tend to pass peak flows and are therefore less likely to detain fine sediment or modify downstream high flows. Alternatively, dams of very different sizes can have similar downstream hydrologic effects depending on how they store and release water over time. However, characterizing dam operations in a meaningful way may be easier for smaller structures (e.g., many of those not included in the national database), because of their smaller storage capacity and limited range of management options.

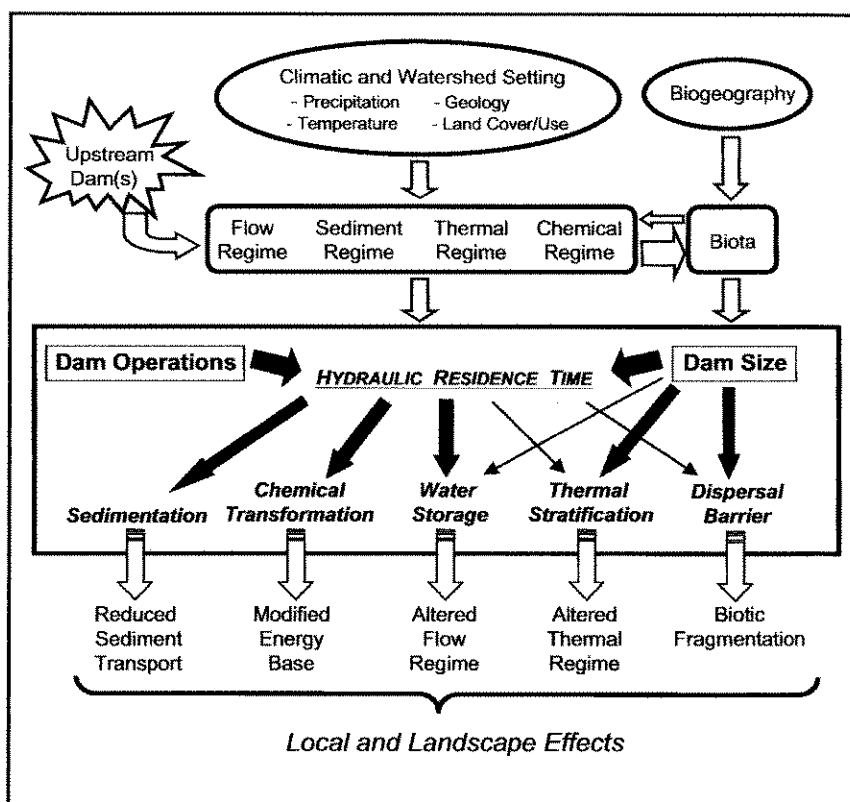


Figure 4. Flow chart illustrating how attributes of dam-reservoir systems, especially dam size and operations, modify fundamental riverine biophysical processes to cause alterations with local and landscape environmental effects.

The influence of dam characteristics on removal decisions

According to a recent compilation, 467 dams have been completely or partially removed in the United States in the 20th century (AR/FE/TU 1999). At least another 30 dams have been completely removed through 2001 (Molly Pohl, Department of Geography, San Diego State University, personal communication, 5 March 2002). What kinds of dams are being removed, and how might future dam-removal decisions be related to variation in dam characteristics?

There are two striking dam-removal patterns: Dams are being removed at an accelerating rate (figure 5a), and the majority of dams being removed are less than 5 m in height (figure 5b). Several factors suggest that small dams will continue to be removed more often than large dams: As indicated by the Wisconsin and Utah databases, dams less than 5 m in height are far more numerous than large dams. Most of these small dams do not generate hydroelectricity or control floods, so the economic benefits of maintaining them are not as great when compared with large dams. Small dams are often older than large dams, which makes it more likely that they will be in poor condition. In fact, concerns about public safety, as well as high repair costs, were major factors affecting decisions to remove a number of old dams (average age > 100 years) in Wisconsin (Born et al. 1998). Small dams are more likely to be abandoned, so that financial burdens asso-

ciated with their safety, repair, and maintenance often fall to local governments and, ultimately, to taxpayers. Indeed, many dams that have been removed were previously abandoned (Shuman 1995). These patterns clearly demonstrate that the current focus on small dam removal is influenced by social and economic factors, as well as by concerns about the environmental effects of small dams (AR/FE/TU 1999, Doyle et al. 2000).

Sediment accumulation in reservoirs is another factor that can influence many dam-removal decisions. This issue can be complicated, depending on the quality and quantity of accumulated sediments, as well as on public and agency attitudes about potential downstream effects of sediment. For example, if toxic contaminants are present in the sediment, there are certain to be concerns about the risks associated with the downstream release of sediments following dam removal, and the potential effects of these sediments on human and ecosystem health (Shuman 1995).

Even when contaminants are absent, accumulated sediment can still influence the likelihood of dam removal. For example, as reservoirs fill with sediment, they often become less effective in controlling floods, storing water, and generating hydropower, which could accelerate calls for dam removal. A

useful index of the operational problems caused by accumulated sediments is the time it takes for 50% of the storage capacity of the reservoir to be lost to sediment deposition (Morris and Fan 1998). The proportional rate at which a reservoir's storage volume fills with sediment depends on basinwide erosion rates (which vary regionally), but also exhibits an inverse relationship to dam size. Empirical data collected for reservoirs across the country by Dendy and colleagues (1973) showed that those having a storage capacity between 1.2×10^6 and 12×10^6 m³ had a median time to half-filling of 91 years (based on data reported in Morris and Fan 1998). Taking the median value of this size range, 92% of the approximately 76,500 dams in the national database are expected to become half-filled with sediment in an average of 91 years. The regional distribution of these short-lived dams varies somewhat (figure 6), with between 74% (California, Nevada) and 94% (Southeast) of dams falling into this category. The age of existing dams also shows regional variation, with as many as 50% of dams having construction dates before 1920 in the Northeast, and as few as 5% in the Plains states.

The extent of this sediment problem is even greater if we consider the estimated 2,000,000 small dams not in the national inventory. These structures are defined as having less than 6.2×10^4 m³ of storage (Graf 1993) and thus would be expected to become half full of sediment within roughly 25 to 40 years. Dendy (1968) estimated that "if present siltation rates continue, about 20% of the Nation's small reservoirs will be half filled with sediment...in about 30 years." The lack of a national database for these structures precludes an estimation of their retirement times. Many in the Northeast are already full of sediment, however (Laura Wildman, American Rivers, Northeast Field Office, personal communication, 22 May 2002), and literally thousands more nationwide will fill in the coming decades. For example, in Wisconsin alone, over 800 dams are less than 2 m high, and about one-third of these were built before 1960. On the basis of the previous estimates, these dams should already have lost more than 50% of their storage capacities.

Sediment accumulation can be a factor that either increases or decreases the likelihood of dam removal, depending in part on local circumstances. For example, in situations where sediment accumulation has reduced the functional ability of dams (e.g., for flood control) and disrupted downstream geomorphic processes, there have been increased calls for dam removal (Matilija Coalition 2000). By contrast, concerns have sometimes been raised about the possibility that downstream habitats, species, and ecosystem processes could be adversely affected (at least in the short term) by the release of large volumes of sediment during dam removal. Mechanized removal before dam breaching is one alternative to sediment release (ASCE 1997), although this can be very expensive. For many of the smaller dams currently being removed, however, the volume of accumulated sediment may be similar to the average annual sediment flux. In these situations, no special management practices are employed, and

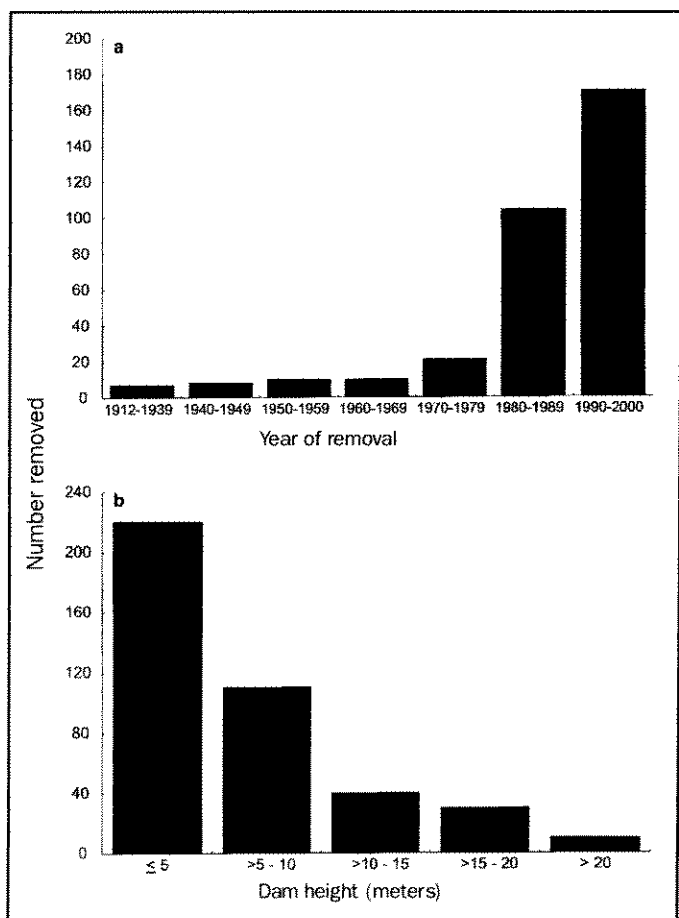


Figure 5. Dam removal in the United States by (a) decade and (b) structure height. (Data taken from AR/FE/TU 1999 and Doyle et al. 2000.)

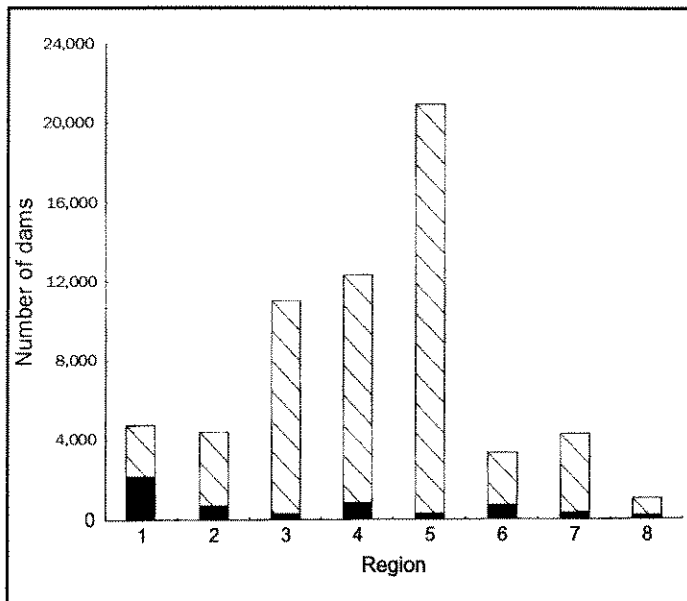


Figure 6. Number of dams by geographic region in the National Inventory of Dams database (USACE 2000) with estimated average time to half-filling with sediment being 91 years or less, on the basis of reservoir storage volumes. Dams are divided into those constructed before (solid black) and after (diagonally hatched) 1920. Refer to figure 2 for regional key.

sediments are allowed to move downstream following dam removal.

The future of dam removals

The rapid aging of dams (especially small ones) and the costs of maintaining old dams practically ensures that dam removal will continue at a brisk pace for the foreseeable future. An open question is whether these removals will be guided by scientific principles aimed at river restoration and conservation or whether they will simply follow utilitarian economic principles (Pejchar and Warner 2001).

In the last decade, an understanding about how dams severely impair free-flowing rivers has become firmly established both in the United States and abroad (Ligon et al. 1995, Collier et al. 1996, NRC 1996, Pringle et al. 2000, WCD 2000). This knowledge has entered into the public debate on river conservation, both in terms of greater willingness of reservoir managers to minimize downstream ecological effects (Muth et al. 2000) and of increased calls for outright dam removal (Pyle 1995, Joseph 1998, AR/FE/TU 1999). These scientific and social currents have led some to call for a new "water ethic" of increasing water-use efficiency through nonstructural means (Gleick 1998, Postel 2000). Such an ethic is needed if human demands for freshwater continue to grow in the coming decades (Postel 2000) and if society wishes to maintain the long-term sustainability of river ecosystems (Naiman and Turner 2000, Baron et al. 2002). The growing pressure for dam removal represents a real opportunity for scientists. Certainly, dam removals provide excellent opportunities for

scientists to perform large-scale experiments in river restoration (Grant 2001, Hart et al. 2002) and thus expand our empirical knowledge base. Moreover, scientists are increasingly likely to be asked to predict the success of dam removal in specific situations where controversy exists over potential benefits and costs. Because dam removal can sometimes be expensive and its ecological effects hard to predict, scientists need to develop a better framework for characterizing dams according to their current environmental effects, as well as to the potential environmental benefits that could accrue following removal. For example, Hart and colleagues (2002) present a graphical model for examining how potential responses to dam removal vary with dam and watershed characteristics. This scientific challenge is made more difficult because the effects of dams result both from their alteration of natural biophysical processes and from human management practices. In this article, we have attempted to highlight some of the more salient attributes of this complex, multidimensional challenge. Developing a more predictive environmental science of dam removal is needed to help society decide where to spend limited resources to maximize restoration potential for impaired river systems in the United States and elsewhere.

Acknowledgments

We thank Dave Pepin for assistance with database analysis and graphics and Chester Watson for providing literature on dam aging. The ideas in this article have benefited greatly from discussions with many colleagues in the Patrick Center's Institute for River Restoration. We thank Matt Greenstone, Laura Wildman, Emily Stanley, and two anonymous reviewers for providing valuable comments on an earlier draft of this manuscript. N. L. P.'s contribution to this project was supported, in part, by grants from the National Science Foundation (# DEB-0075352) and the US Environmental Protection Agency (SPO # BS0056363). D. D. H.'s work was partially supported by the Patrick Center Endowment Fund. Completion of our manuscript was also facilitated by support from The Pew Charitable Trusts.

References cited

- [AR/FE/TU] American Rivers, Friends of the Earth, Trout Unlimited. 1999. Dam Removal Success Stories: Restoring Rivers through Selective Removal of Dams That Don't Make Sense. Washington (DC): AR/FE/TU.
- [ASCE] American Society of Civil Engineers. 1997. Guidelines for retirement of dams and hydroelectric facilities. New York: American Society of Civil Engineers.
- . 2001a. Web Pages about Dam Infrastructure Report Card. Reston (VA): ASCE. (5 October 2001; www.asce.org/reportcard/index.cfm?reaction=factsheet&page=8)
- . 2001b. Web Pages about Issue Brief on Dams. Reston (VA): ASCE. (12 November 2001; www.asce.org/news/reportcard/issbrfdam.cfm)
- Babbitt B. 2002. What goes up, may come down. *BioScience* 52: 656–658.
- Baron JS, Poff NL, Angermeier PL, Dahm CN, Gleick PH, Hairston NG, Jackson RB, Johnston CA, Richter BG, Steinman AD. 2002. Meeting ecological and societal needs for freshwater. *Ecological Applications* 12. Forthcoming.
- Bayley PB. 1995. Understanding large river-floodplain ecosystems. *BioScience* 45: 153–158.

- Benke AC. 1990. A perspective on America's vanishing streams. *Journal of the North American Benthological Society* 9: 77-88.
- Born SM, Genskow KD, Filbert TL, Hernandez-Mora N, Keefer ML, White KA. 1998. Socioeconomic and institutional dimensions of dam removal: The Wisconsin experience. *Environmental Management* 22: 359-370.
- Chao BF. 1991. Man, water and global sea level. *EOS, Transactions, American Geophysical Union* 72: 492.
- Collier M, Webb RH, Schmidt JC. 1996. Dams and rivers: Primer on the downstream effects of dams. Reston (VA): US Geological Survey. Circular no. 1126.
- Dendy FE. 1968. Sedimentation in the nation's reservoirs. *Journal of Soil and Water Conservation* 23: 135-137.
- Dendy FE, Champion WA, Wilson RB. 1973. Reservoir sedimentation surveys in the United States. Pages 349-357 in Ackermann WC, White GF, Worthington EB, eds. *Man-Made Lakes: Their Problems and Environmental Effects*. Washington (DC): American Geophysical Union. Geophysical Monograph 17.
- Doyle MW, Stanley EH, Luebke MA, Harbor JM. 2000. Dam removal: Physical, biological and societal considerations. Paper presented at the American Society of Civil Engineers Joint Conference on Water Resources, Engineering and Water Resources Planning and Management; 30 July-2 August; Minneapolis, MN.
- EnergyIdeas. 2001. Energy glossary: Hydroelectric. Washington State University Cooperative Extension Energy Program, Energy Ideas Clearinghouse. (1 July 2002; www.energyideas.org/glossary/terms.cfm?categoryID=7)
- [EPA] US Environmental Protection Agency. 2001. Dams management measures. EPA Office of Water. (1 July 2002; www.epa.gov/nps/MMGI/Chapter6/ch6-3a.html)
- Fausch KD, Torgersen CE, Baxter CV, Li HW. 2002. Landscapes to riverscapes: Bridging the gap between research and conservation of stream fishes. *BioScience* 52: 483-498.
- [FEMA] Federal Emergency Management Agency. 2001. National Dam Safety Program. FEMA Federal Insurance and Mitigation Administration. (1 July 2002; www.fema.gov/nit/damsafe/about.htm)
- Gleick PH. 1998. *The World's Water: The Biennial Report on Freshwater Resources*. Washington (DC): Island Press.
- Graf WL. 1993. Landscapes, commodities, and ecosystems: The relationship between policy and science for American rivers. Pages 11-42 in *Water Science and Technology Board, National Research Council. Sustaining Our Water Resources*. Washington (DC): National Academy Press.
- . 1999. Dam nation: A geographic census of American dams and their large-scale hydrologic impacts. *Water Resources Research* 35: 1305-1311.
- Grant G. 2001. Dam removal: Panacea or Pandora for rivers? Invited commentary. *Hydrologic Processes* 15: 1531-1532.
- Hart DD, Johnson TE, Bushaw-Newton KL, Horwitz RJ, Bednarek AT, Charles DF, Kreeger DA, Velinsky DJ. 2002. Dam removal: Challenges and opportunities for ecological research and river restoration. *BioScience* 52: 669-681.
- Joseph P. 1998. The battle of the dams: Those who think some of our rivers are a dammed shame argue for the structures to come down. *Smithsonian* (November).
- Kalff J. 2002. *Limnology: Inland Water Ecosystems*. Upper Saddle River (NJ): Prentice Hall.
- Ligon FK, Dietrich WE, Trush WJ. 1995. Downstream ecological effects of dams. *BioScience* 45: 183-192.
- Matilija Coalition. 2000. *Matilija Coalition*. (1 July 2002; www.rain.org/~pijenkin/matilija)
- McCully P. 1996. *Silenced Rivers: The Ecology and Politics of Large Dams*. London: Zed Books.
- Meade RH, ed. 1995. *Contaminants in the Mississippi River, 1987-1992*. Reston (VA): US Geological Survey. Circular no. 1133.
- Morris GL, Fan J. 1998. *Reservoir Sedimentation Handbook: Design and Management of Dams, Reservoirs, and Watershed for Sustainable Use*. New York: McGraw-Hill.
- Muth RT, Crist LW, LaGory KE, Hayse JW, Bestgen KR, Ryan TP, Lyons JK, Valdez RA. 2000. *Flow and Temperature Recommendations for Endangered Fishes in the Green River Downstream of Flaming Gorge Dam*. Lakewood (CO): Upper Colorado River Endangered Fish Recovery Program. Final Report, Upper Colorado River Endangered Fish Recovery Program Project FG-53.
- Naiman RJ, Turner MG. 2000. A future perspective on North America's freshwater ecosystems. *Ecological Applications* 10: 958-970.
- Nilsson C, Berggren K. 2000. Alterations of riparian ecosystems caused by river regulation. *BioScience* 50: 783-792.
- [NRC] National Research Council. 1996. *Upstream: Salmon and Society in the Pacific Northwest*. Washington (DC): National Academy Press.
- Pejchar L, Warner K. 2001. A river might run through it again: Criteria for consideration of dam removal and interim lessons from California. *Environmental Management* 28: 561-575.
- Pennsylvania Fish and Boat Commission. 2001. *Hazards on the water*. Commonwealth of Pennsylvania Fish and Boat Commission (1 July 2002; http://sites.state.pa.us/PA_Exec/Fish_Boat/damhaz.htm)
- Petts GE. 1984. *Impounded Rivers: Perspectives for Ecological Management*. New York: John Wiley and Sons.
- Poff NL. 1996. A hydrogeography of unregulated streams in the United States and an examination of scale-dependence in some hydrological descriptors. *Freshwater Biology* 36: 71-91.
- Poff NL, Allan JD, Bain MB, Karr JR, Prestegard KL, Richter BD, Sparks RE, Stromberg JC. 1997. The natural flow regime: A paradigm for river conservation and restoration. *BioScience* 47: 769-784.
- Postel SL. 2000. Entering an era of water scarcity: The challenge ahead. *Ecological Applications* 10: 941-948.
- Pringle CM, Freeman MC, Freeman BJ. 2000. Regional effects of hydrologic alterations on riverine macrobiota in the New World: Tropical-temperate comparisons. *BioScience* 50: 807-823.
- Pyle MT. 1995. Beyond fish ladders: Dam removal as a strategy for restoring America's rivers. *Stanford Environmental Law Journal* 14: 97-143.
- [RAW/TU] River Alliance of Wisconsin, Trout Unlimited. 2000. *Dam Removal: A Citizen's Guide to Restoring Rivers*. Madison (WI): River Alliance of Wisconsin.
- Rosenberg DM, McCully P, Pringle CM. 2000. Global-scale environmental effects of hydrological alterations: Introduction. *BioScience* 50: 746-751.
- Sahagian D. 2000. Global physical effects of anthropogenic hydrological alterations: Sea level and water redistribution. *Global and Planetary Change* 25: 39-48.
- Schnitter NJ. 1994. *A History of Dams: The Useful Pyramids*. Rotterdam (The Netherlands): A. A. Balkema.
- Scott ML, Friedman JM, Auble GT. 1996. Fluvial processes and the establishment of bottomland trees. *Geomorphology* 14: 327-339.
- Shuman JR. 1995. Environmental considerations for assessing dam removal alternatives for river restoration. *Regulated Rivers: Research and Management* 11: 249-261.
- Smith N. 1971. *A History of Dams*. London: Peter Davies.
- St. Louis VL, Kelly CA, Duchemin E, Rudd JWM, Rosenberg DM. 2000. Reservoir surfaces as sources of greenhouse gases to the atmosphere: A global estimate. *BioScience* 50: 766-775.
- [USACE] US Army Corps of Engineers. 2000. *National inventory of dams*. CD-ROM Federal Emergency Management Agency. (1 July 2002; <http://crunch.tec.army.mil/nid/webpages/nid.cfm>)
- [USBR] US Bureau of Reclamation. 2001. *Web pages about definition of technical terms*. (1 July 2002; www.usbr.gov/dams/glossary.html)
- Ward JV, Stanford JA, eds. 1979. *The Ecology of Regulated Streams*. New York: Plenum Press.
- Ward JV, Stanford JA. 1983. The serial discontinuity concept of lotic ecosystems. Pages 29-42 in Fontaine TD, Bartell SM, eds. *Dynamics of Lotic Ecosystems*. Ann Arbor (MI): Ann Arbor Science.
- [WCD] World Commission on Dams. 2000. *Dams and Development: A New Framework for Decision-making*. London: Earthscan Publications.
- [WDNR] Wisconsin Department of Natural Resources. 1995. *Wisconsin's Biodiversity as a Management Issue: A Report to the Department of Natural Resources Managers*. Madison (WI): WDNR.
- Wootton JT, Parker MS, Power ME. 1996. Effects of disturbance on river food webs. *Science* 273: 1558-1561.

A Framework for Estimating the Costs and Benefits of Dam Removal

ED WHITELAW AND ED MACMULLAN

Although dams provide a variety of economic goods and services, including electric power, flood control, water supply, reservoir recreation, and navigational services, they also have detrimental effects on riverine ecosystems (Petts 1984). As a result, many people want to know the socioeconomic and ecological benefits and costs of rehabilitating or restoring rivers through dam modification or removal (AR/FE/TU 1999).

Cost-benefit analysis is one economic tool that helps decisionmakers choose among policy alternatives (Boardman et al. 1996). Ideally, cost-benefit analysis includes all of the costs and benefits associated with each policy alternative. In fact, however, costs and benefits can be difficult to measure—estimating the value of an endangered species, for example—or may not be fully recognized at the time a study is conducted. Thus using cost-benefit analysis in evaluating the removal of a dam can challenge even seasoned analysts.

In spite of these limitations, decisionmakers and stakeholders frequently rely on cost-benefit analysis for insights into the potential consequences of modifying or removing dams. The Edwards Dam on the Kennebec River in Maine, removed in 1999, illustrates the point. A cost-benefit study concluded that necessary structural repairs would have cost 1.7 times the cost of removing the dam and restoring anadromous fish passages (AR/FE/TU 1999).

Edwards Dam is small compared with other dams recently under consideration for removal. The US Army Corps of Engineers completed a draft cost-benefit analysis of a proposal to remove a series of four large dams on the lower Snake River in the Pacific Northwest. Wild salmon stocks have dipped perilously low on the river, and many people believe the costs of keeping the dams outweigh the benefits.

In this article, we describe principles we believe are effective in assessing the economic consequences of environmental management decisions. We then describe how those principles might be used for a cost-benefit analysis regard-

SOUND COST-BENEFIT ANALYSES OF REMOVING DAMS ACCOUNT FOR SUBSIDIES AND EXTERNALITIES, FOR BOTH THE SHORT AND LONG RUN, AND PLACE THE ESTIMATED COSTS AND BENEFITS IN THE APPROPRIATE ECONOMIC CONTEXT

ing dam removal using the dams on the lower Snake River as a case study. We examine parts of the US Army Corps of Engineers' draft cost-benefit analysis for these dams and suggest modifications to the Corps' analysis that would more fully account for relevant costs and benefits.

Analytical principles

On 9 September 1998, 78 economists sent a letter to the governors of the four Pacific states and the premier of British Columbia, urging them "to consider the full range of economic consequences" when they and members of their administrations make salmon-management decisions (Whitelaw et al. 1998). Box 1 presents the six principles that the economists emphasized should guide an assessment or cost-benefit analysis of the economic consequences of practically any environmental management decision, including whether to keep or remove a dam.

Ed Whitelaw (e-mail: whitelaw@eugene.econw.com) is a professor of economics at the University of Oregon, Eugene, OR 97403, and president of ECONorthwest, an economic consulting firm with offices in Eugene and Portland, OR, and Seattle, WA. Ed MacMullan is a project manager and economist at ECONorthwest. © 2002 American Institute of Biological Sciences.

Box 1. Six principles that should guide the analysis of the economic consequences of removing or keeping a dam. From a 1998 letter by concerned economists (Whitelaw et al.) to Governors Kitzhaber, Knowles, Locke, and Wilson and Premier Clarke regarding the economic issues of salmon recovery.

Primary analytical principles

1. Benefits as well as costs

Removing or keeping a dam would generate economic benefits as well as economic costs. Consider them both to understand the full effect on the value of the goods and services derived from streams, forests, and other resources.

2. Positive as well as negative impacts on jobs

Dealing with a dam would have both positive and negative effects on job opportunities. Consider them both to understand the full effect on workers, their families, and their communities.

Secondary analytical principles

3. Distribution of consequences and fairness

Those who enjoy the benefits or jobs of a decision on a dam would not necessarily be the same as those who would bear the costs or job losses. Consider the full distribution of economic consequences to understand who wins, who loses, and the fairness of the distribution.

4. Rights and responsibilities

With any decision on a dam, property owners and resource users behave differently than they otherwise would. Consider whether these changes represent infringement of their rights or enforcement of their responsibilities.

5. Uncertainty and sustainability

Any decision on a dam would rely unavoidably on information insufficient to guarantee the outcome. Consider fully the potentially high costs from decisions yielding undesirable outcomes that are irreversible or extremely difficult to reverse.

6. More than just salmon conservation

Removing or keeping a dam would have a variety of ecological and economic effects, such as changes in the quality of stream water used for other purposes, that may seem peripheral. But consider all the effects.

Two of the six principles play primary roles by addressing the two key effects of a decision on a dam: (1) the effect of the decision on the value of the goods and services derived from the environmental resources; and (2) the effect on jobs and associated variables, such as incomes and the well-being of communities. The other four play secondary roles, offering guidance on the issues that should be addressed when applying the first two principles. The four secondary principles are just as important as the primary ones, but they play a different role, defining the range of issues that should be taken into account as one looks at the benefits, costs, and effects on jobs.

The first principle—first in order and first in priority—admonishes decisionmakers to consider both the benefits and the costs. Though this may seem eminently reasonable, the economists observed that many economic studies of environmental management decisions predominantly emphasize the costs (Whitelaw et al. 1998). Doing so reduces the perceived economic importance of the environmental resources. When weighing the benefits and costs, decisionmakers should take into account how their decision would affect all goods and services with economic value, not just those traded in markets with monetary prices. In addition, a full accounting must be provided of the true value of each affected good or service, taking into account the market price, as well as all factors, such as subsidies, taxes, and environmental externalities, that distort the level of supply or demand. (Environmental externalities occur, for example, when those who generate pollution benefit financially while others downstream or down-

wind pay the costs of the pollution.) Finally, the estimates of economic impacts—costs, benefits, employment consequences, and so on—should be placed in the context of the size and makeup of local and regional economies. Considering impacts without the proper context limits the usefulness of information to decisionmakers or stakeholders.

A decision to remove a dam also would have both positive and negative effects on jobs and incomes. When examining these effects, decisionmakers should take into account the economy's ability to adjust over time to exploit the positive and attenuate the negative. The decision to remove a dam may expand or contract the demand for labor and, hence, increase or decrease job opportunities. The actions taken may affect the quality of life in a local area or region and, hence, influence where people prefer to live, work, play, and shop. Analysis of the employment effects associated with a management decision on a dam must also separate these consequences from the employment consequences associated with larger economic forces and trends *unrelated* to decisions on the dam, but which may affect local and regional economies in proximity to the dam.

Because the decision on a dam generates both benefits and costs, and produces both positive and negative effects on jobs and incomes, it creates both winners and losers. The economists recommended that such distributional effects not be overlooked. They also emphasized the importance of having a clear understanding of how the decision affects the rights and responsibilities of landowners and resource users. The

value society places on the decision that restricts property owners' rights can differ markedly from the value of otherwise comparable measures that induce the property owners to comply with their responsibilities. In addition, the economists observed that, given the uncertainty regarding a decision to remove a dam, there always is the possibility it would yield undesired outcomes, and care should be taken to avoid outcomes that are costly—or even impossible—to reverse. Finally, the economists stressed that although the primary economic consequences of an environmental management decision have to do with the specific environmental resource itself, others do not. A full analysis should include the costs, benefits, and effects peripheral to the specific resource at issue.

The six principles identified in the economists' letter provide an analytical foundation for assessing the costs and benefits of removing a dam. The analytical approach outlined in this section describes the economic implications of removing a dam, from large dams that facilitate barge traffic to small dams such as those that provided water to long-abandoned mills. Removing larger dams is likely to generate more significant impacts than removing smaller dams; therefore, depending on the specifics of the dam, a full-blown economic analysis may not be appropriate. (See Trout Unlimited 2001 for information on the economic benefits of removing small dams.) However, a cost-benefit analysis of removing a dam using the principles described in box 1 ensures that the analysis captures the full range of economic consequences.

Application to dams on the lower Snake River

In this section we describe the application of the analytical approach described above to the question of removing the four dams on the lower Snake River in Washington.

Background regarding the impacts of dams on endangered salmon.

Four dams are situated in the lower Snake River, between the Snake's confluence with the Columbia River at Pasco, Washington, and Lewiston, Idaho. The US Army Corps of Engineers (USACE) constructed the dams between 1962 and 1975, primarily to create a series of ponds so barges could reach Lewiston, and secondarily to provide easy access to water for irrigation and to generate hydroelectricity.

Wild salmon stocks returning to the Snake River have plummeted since the dams' construction, and a chorus of fisheries biologists and others has called for breaching or bypassing the dams, that is, removing the earthen mounds adjacent to the concrete portions of the dams and letting the rivers run free. Proponents argue that breaching the dams would, among other things, restore endangered wild salmon, return traditional sites and fisheries to Indian tribes, improve water quality, reduce taxpayer subsidies to corporate irrigators and barging companies, and comply with numerous laws and treaties. Opponents claim such actions would prove prohibitively costly, even wreck the Northwest's economy.

In December 1999, the US Army Corps of Engineers released a draft Feasibility Report/Environmental Impact Statement (FR/EIS), which, among other things, provides an estimate of the economic effects of breaching the four dams on the lower Snake River (USACE 1999a). The FR/EIS describes, to varying degrees, the costs and benefits of dam removal on different sectors of the regional and national economies, including tribal interests; recreational use; anadromous fisheries; irrigated agriculture; transportation; electrical utilities; and municipal, industrial, and private water use.

In the remainder of this article, we evaluate the Corps' analysis of the economic effects of breaching the dams in light of the analytical principles described in box 1. We have limited our critique of the Corps' analysis to the primary analytical principles of the overall costs and benefits of removing the dams and the associated impacts on jobs. Readers interested in a more in-depth discussion of the primary analytical principles and the related secondary principles as they apply to the Corps' cost-benefit analysis should consult ECONorthwest (1999, 2000).

Evaluating the Corps' analysis of economic effects.

First we describe the overall structure of the Corps' cost-benefit analysis, and then we review the Corps' analysis of costs and benefits. This section concludes with our critique of the Corps' analysis of the employment impacts of removing the dams.

The Corps' overall analytical approach. When federal agencies such as the Corps conduct cost-benefit analyses of proposed water projects, they typically follow the Principles and Guidelines developed by the US Water Resources Council (USWRC) in the early 1970s to provide guidance on decisionmaking and analytical procedures as they apply to water resources. The Principles and Guidelines, which replaced the Principles and Standards, were last updated in 1983 (USWRC 1983). Other federal agencies that use the Principles and Guidelines are the Bureau of Reclamation, the Natural Resources Conservation Service, and the Tennessee Valley Authority (NRC 1999). According to the Corps (USACE 1999b), the Principles and Guidelines recommend that a cost-benefit analysis include the following socioeconomic factors:

- National economic development (NED) effects, which describe the changes in the economic value of the national output of goods and services
- Environmental quality effects, which describe nonmonetary consequences for significant natural and cultural resources
- Regional economic development (RED) effects, which address changes in the distribution of regional economic activity such as jobs and income
- Other social effects, which describe potential effects from relevant perspectives that are not reflected in the other three types of effects



In spite of the comprehensive approach outlined in the Principles and Guidelines, the Corps considered only a portion of this information in its decisionmaking process for the dams on the lower Snake. The Corps' FR/EIS states, "The NED account is the only account required under the WRC [Water Resources Council] guidelines" (appendix I, pp. 11-1-11-2). As calculated by the Corps, the impact of dam removal on the value of the nation's goods and services apparently determined the outcome of its cost-benefit analysis.

The National Research Council (NRC) reviewed the Corps' use of the Principles and Guidelines in a number of applications and concluded that the Corps' approach ignores important impacts, is out-of-date, and does not reflect current thinking on the role that water resources play in local, regional, or national economies (NRC 1999). The NRC concluded (pp. 4-5):

While they were in effect, the P&S [Principles and Standards] were consistently reviewed and updated by federal and other water planning specialists. By contrast, the P&G [Principles and Guidelines] have not received the same degree of attention and, as a result, do not adequately reflect contemporary water resource planning principles and practices.... Movement away from consideration of the National Economic Development (NED) account [is] the most important concern. Today, ecological and social considerations are often of great importance in project planning and should not necessarily be considered secondary to the maximization of economic benefits. Strict adherence to the NED account may discourage consideration of innovative and non-structural approaches to water resources planning.... In summary, the committee recommends that the federal Principles and Guidelines be thoroughly reviewed and modified to incorporate contemporary analytical techniques and changes in public values and federal agency programs.

Applying the NRC's criticisms of the Corps' overall analytical approach to the analysis of impacts of removing the Snake River dams, we see that the Corps' analysis provides limited useful information and misleading results. For example, as we describe below, the Corps' NED analysis is incomplete because, among other deficiencies, it excludes the impact of subsidies. Thus the Corps violated the first principle in box 1 of considering the full range of economic benefits and costs. Even though the Corps estimated RED impacts, they ignored these impacts during the decisionmaking process and focused exclusively on the NED impacts. By excluding RED impacts from the decisionmaking process, they violated the second principle of considering the full range of employment impacts. Likewise, the Corps excluded the range of secondary analytical principles in box 1 from its analysis.

Changes in benefits and costs. The Corps' analysis of costs and benefits—the NED effects described in the Principles and Guidelines, and the effects that drove the Corps' decisionmaking process—has significant deficiencies. We first discuss two of the major drawbacks of the Corps' analysis: ex-

cluding subsidies from the cost-benefit analysis and ignoring benefits associated with tribal circumstances and non-market values. The former overestimates the cost of taking out the dams. The latter underestimates the benefits of taking out the dams. We then discuss the Corps' results and place these results in the context of the regional economy of Washington, Oregon, and Idaho.

Certain sectors of the region's economy that rely on the dams benefit from subsidies provided by the federal government. In effect, taxpayers throughout the United States subsidize the economic activities and profits of these businesses. Taking out the dams would generate negative economic consequences for these businesses, but there are positive economic consequences for US taxpayers because they would no longer pay subsidies to these businesses.

Transportation is one example. Snake River waterway users pay a fuel tax that generates a few hundred thousand dollars annually, which covers but a small portion of the actual costs of using the waterways. Federal taxpayers make up the difference, contributing \$10 million annually to subsidize transportation's share of operations and maintenance costs for the Snake River dams (Dickey 1999). The Corps did not include this and other subsidies in their analysis. For example, describing the analysis of transportation impacts, the FR/EIS states, "The analysis does not take into consideration the effects of taxes or subsidies, which represent transfer payments within the national economy" (USACE 1999b, p. 13-62). Furthermore, subsidies are more wide-reaching than simply transferring wealth from one group to another. When a service, such as transportation along the Snake River, is subsidized, so that users do not face a price reflecting the full production cost, they have an economic incentive to consume more of the service than they would otherwise. The Corps' cost-benefit analysis failed to account for this overconsumption, which biases the analysis in favor of those sectors of the economy that receive subsidies.

In estimating the benefits from breaching the dams, the Corps excluded a number of relevant values, including tribe-related benefits and the benefits that all of us gain from the existence of both the increased salmon runs and a free-flowing lower Snake River. First, the Corps' estimate of tribe-related benefits included the number of acres of sacred and traditional sites that the tribes would regain access to, as well as the number of pounds of fish from treaty-protected subsistence and ceremonial fisheries, but it did not include the economic benefits that tribal members and other Northwesterners and Americans would gain from these changes (USACE 1999b). In not doing so, it overlooked economic benefits to tribal members that constitute real increases in the value of national goods and services. As a result, the Corps underestimated how breaching the dams would benefit the tribes, and how that, in turn, would benefit all of us.

Second, the Corps excluded from its cost-benefit tally what it calls passive-use benefits that Northwesterners and other Americans would enjoy from both the increased salmon runs and from converting the lower Snake to 140 miles of free-

flowing river. These values come not from using the resources—the salmon or the river—but from knowing the salmon and the free-flowing river exist and that future generations would get to enjoy them. These values aren't trivial. Economists working for the Corps estimated that the passive-use values of these two resources range from \$486 million to nearly \$1.3 billion (USACE 1999b). The Corps' estimates of the overall costs and benefits of taking out the dams ranged from a net cost of \$300 million to a net benefit of \$1.3 billion, in 1998 dollars (Whitelaw 2000). If passive-use values were incorporated into the Corps' overall estimate of net costs and benefits, the range would change to a low of \$186 million net benefits and a high of \$2.6 billion net benefits. This huge range stems largely from the wide range in the estimates of the benefits from river recreation that would result from breaching the dams, a range of \$11 million to \$1.5 billion (USACE 1999b).

Personal income provides one measure of the ability of a region to pay for some good, service, or action. In this case it serves as a context for the Corps' cost-benefit results. Comparing the region's personal income with the net costs or benefits of taking out the dams provides insights into the relative expense or benefit of the action. Personal income in Oregon, Washington, and Idaho in 2000 exceeded \$310 billion (USDC 2001). The Corps' worst-case estimate of net costs of \$300 million represents 0.1% of the region's personal income. The actual impacts would be even smaller because, as we described above, the Corps overestimated the costs and underestimated the benefits of taking out the dams.

Employment impacts. In spite of the Corps' emphasis on changes in costs and benefits at the national level, the impacts on jobs, especially jobs in the local and regional economy, are what concern many people. We discuss how the Corps' analysis overstates the employment impacts and then place the Corps' analytical results in the proper context, in this case, the local and regional economies that would be affected by the decision. We also illustrate the importance of considering relevant economic forces and trends in an analysis of employment impacts.

Figure 1 summarizes the Corps' estimates of the effects that breaching the dams would have on jobs, that is, the regional economic development effects. (See ECONorthwest 1999 and Whitelaw 2000 for a detailed discussion of the employment impacts of bypassing the dams.) According to the

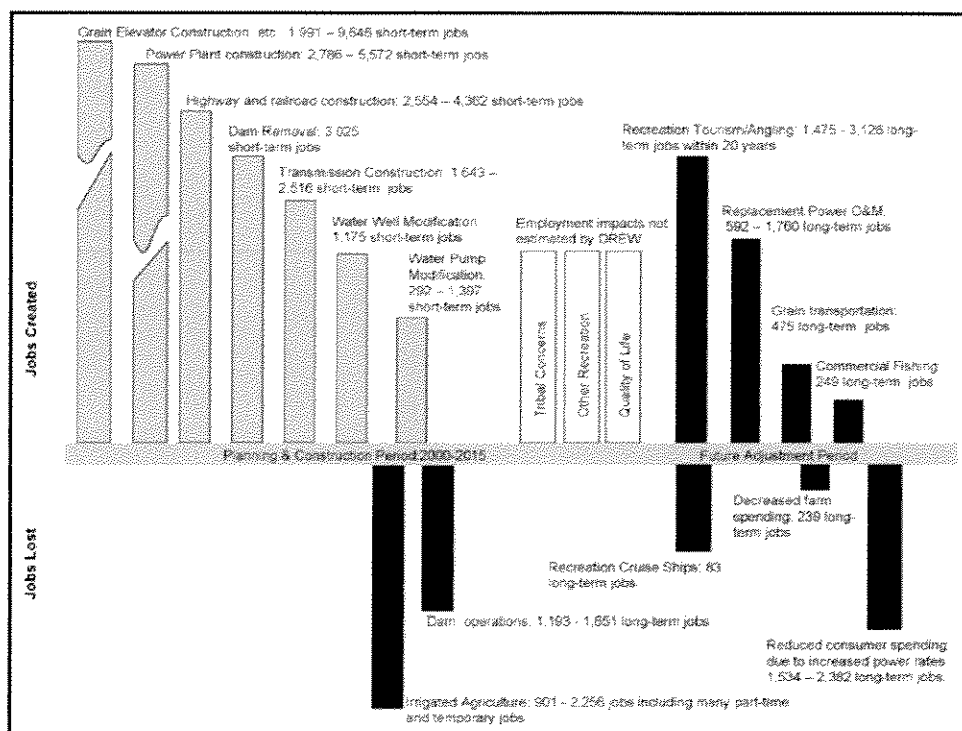


Figure 1. Employment impacts of the bypass. Adapted by ECONorthwest with data from USACE (1999b).

Corps, breaching the dams would create 13,400 to 27,700 short-term jobs during the decade of deconstruction and construction (USACE 1999b).

In the long term, there would be job losses and gains. The biggest gains, between 1475 and 3126 jobs, would result from improved recreational tourism and angling opportunities. The largest losses of long-term jobs would occur in irrigated agriculture (between 901 and 2256 jobs), the operations of the existing dams and locks (between 1193 and 1651), and reduced spending caused by increased electricity rates (between 1534 and 2382). Taking the midpoints of the ranges shown in figure 1, breaching the dams would cause a net loss of 1081 long-term jobs in the Pacific Northwest (4200 jobs gained, 5281 jobs lost). The Corps generated ranges of employment impacts by using low, medium, and high scenarios for various data and assumptions.

The Corps overestimated the negative employment consequences of bypassing the dams because it failed to account for the economic forces and trends acting on the relevant economies. The tool the Corps employed to estimate the impact of breaching on jobs and incomes "presents a picture of the economy at a single point in time," the Corps states, and that point is 1995 (USACE 1999b, p. I6-5). Furthermore, the Corps assumes "the long-run effects are permanent and continue for the 100-year period analyzed in this study" (USACE 1999b, p. I6-3). In other words, the Corps assumed that the basic structure of the economy would remain fixed in its 1995 form, unchanged for the next 100 years. For example, the Corps estimated a maximum of 2256 jobs would be lost in irrigated agriculture (figure 1). To arrive at 2256, the Corps

assumed that, when breaching the dams eliminates reservoir water for irrigation, the affected 13 corporate farms would take out of production all 37,000 acres of their farmland. This assumption ignores other possible outcomes, including switching to groundwater, adopting different irrigation practices, and altering crops. In effect, the Corps assumed that the owners of these corporate assets would quit and the assets would remain idle for 100 years. Furthermore, the Corps assumed, in effect, that for the next century, those who lost their jobs as a result would never work again; local and regional firms that otherwise would have sold goods and services to those who lost their jobs instead would lose those sales and wouldn't find replacement sales; owners of the farming enterprises wouldn't switch to any other economic activities; and those throughout the chain who lost their jobs would act exactly the same way as the original job losers in that they would never work again.

The Corps' rigid analytical structure produces an extreme worst-case scenario, unsupported by economic theory or by the historical performance of the local and regional economies. The Corps' analysis freezes all economic interactions in 1995. Such a constraint ignores the dynamic adjustments that economies—employees and employers, buyers and sellers, savers and investors, and all other economic decisionmakers—undertake all the time. For example, since the four dams began service, the agricultural sector experienced four major contractions, each of which affected more than 2256 workers, the maximum that breaching the dams would affect, the Corps predicts. And yet the local and regional economies have expanded steadily during this period (USDC 1998, ECONorthwest 1999).

These data and economic trends indicate that a snapshot of lost jobs at a point in time tells very little about how a real economy reacts to the breaching of four dams or any other changes. Such rigid and unrealistic assumptions cannot produce a credible forecast of economic consequences under the breaching scenario. University of Montana economist Tom Power equates such an analytical approach to driving by looking in the rearview mirror (Power 1996).

A comprehensive assessment of likely employment consequences of bypassing dams would include these elements:

- Feasible alternatives to permanently idling assets that are negatively affected by the bypass
- Information on the average periods of unemployment in the local and regional economies
- Likely mitigation options that would reduce negative employment consequences
- Projected employment demand in economic sectors unaffected by the bypass

It is instructive to put these estimates in perspective. By the end of 2000, Washington, Oregon, and Idaho had approximately 6.2 million workers (USDC 2001). For the three states, the net loss of 1081 jobs would amount to less than 0.02% of

all jobs. For the counties in southeastern Washington, north-eastern Oregon, and central Idaho near the lower Snake River—the counties the Corps treated as the relevant local economy—the Corps estimated a net loss of 711 long-term jobs, less than 0.3% of the employment in these 15 counties. (In 1996, the local economy of the 15 counties that border the lower Snake River employed approximately 266,000 workers.) For another perspective, compare the total number of jobs the Corps predicted would be lost—5281 gross, not net—with the 25,000 jobs lost in Oregon and Washington's timber industry during the past decade (USDC 2000).

Placing the Corps' results regarding employment impacts in the context of the size of the local and regional economies (and ignoring the issues raised by the NRC) indicates that bypassing the dams would generate minimal negative employment consequences relative to the size of the local and regional economies. Even though the negative employment impacts would be minimal overall, they represent hardships for the affected workers and their families. The limited nature of the negative impacts, however, means that mitigating the negative employment consequences would be manageable. (For more information on mitigation options, see ECONorthwest 1999.)

Discussion and conclusions

In a 27 July 1999 speech, Senator Slade Gorton (R-WA) claimed that removing the four Snake River dams would be an "unmitigated disaster and an economic nightmare" (Hughes 1999). In February 2000, George W. Bush said, "Breaching the [Snake River] dams would be a big mistake....The economy and jobs of much of the Northwest depend on the dams" (*Seattle Times*, 26 February 2000, p. A1). In its 1 May 2000 editorial, the *Oregonian* likened breaching the dams to "taking a sledgehammer to the Northwest economy." The Clinton administration, perhaps sensitive to these claims, decided to leave the dams in place while other salmon-recovery methods were attempted.

Just 10 years ago, many politicians offered similar predictions on the disastrous effects of protecting the northern spotted owl. Representative Bob Smith (R-OR) predicted the owl listing would "wreak havoc on the people and economy of the Pacific Northwest" (Ulrich and Ota 1990). During a campaign swing through the Pacific Northwest in 1992, President George Bush warned, "It is time we worried not only about endangered species, but endangered jobs" (Hong and Yang 1992). President Bush and many of the other politicians in those years—Senators Mark Hatfield and Bob Packwood and Representative Bob Smith—embraced the simplistic logic of owls versus jobs, just as some today frame the dam-breaching debate as salmon versus jobs: We can protect endangered jobs, or we can protect endangered species, but not both.

In fact, the Pacific Northwest economy has boomed, consistently outperforming the national economy, whether measured by jobs, income, or sheer exuberance, throughout the 1990s. Between 1988 and 1998, logging in Oregon and Wash-

ington fell 91% on federal lands and 52% overall, and timber-industry employment dropped 20%. But new jobs in other sectors offset these losses. Total employment actually increased 31%, while inflation-adjusted per capita income grew 26% (USDC 2000, Warren 1990–2000). Ten years ago this region had never experienced widespread economic changes to protect a species. The current deliberations on the fate of the Snake River dams would benefit from a consideration of these experiences and the implications for developing and conducting cost–benefit analyses. Ignoring the constraints inherent in the Corps' analysis and the resulting biases that overestimate costs and underestimate benefits, the jobs losses predicted by the Corps do not describe a “disaster,” “nightmare,” or “sledgehammer” for either the local or the regional economy.

An incomplete or otherwise flawed analysis lends itself to such misrepresentation. It also fails to characterize accurately the range of potential economic consequences. By applying the analytical principles in box 1, however, cost–benefit analysts would meet the relevant professional standards and, not incidentally, provide more useful information to decisionmakers. For emphasis and clarification, we describe below the important components of the two primary analytical principles.

Measure all relevant costs, benefits, and employment gains and losses. A policy decision will rarely, if ever, generate only costs or only benefits. The impacts of removing dams, for example, extend far beyond dams, fish, and farmers, just as the Pacific Northwest found that the impacts of restricting logging extended far beyond owls and timber workers. In some cases a policy decision may generate costs or benefits some distance away from the area directly affected by the decision. For example, removing a dam may influence populations of anadromous fish, which in turn influences incomes and employment far downstream in coastal communities engaged in commercial fishing.

Account for all costs and benefits including subsidies and externalities. Ignoring the subsidies to the transportation sector overestimates the true costs of the bypass. To the extent that the existence and operation of a dam generates negative externalities, such as raising the temperature of the water, removing a dam yields benefits.

Place the estimated costs and benefits in the appropriate context. In this case, the context for an analysis of removing the dams is the same as it was for protecting the spotted owl: the local and regional economies affected by the decision.

The goal of cost–benefit analyses is providing decisionmakers and others with useful information on the range of likely economic consequences of policy decisions. As with other analytical efforts, the overall structure of the analysis influences the extent to which this goal is achieved. We believe that the set of principles outlined in this article provide a sound framework for estimating the costs and benefits of removing dams.

Acknowledgments

We thank Angela Bednarek for her valuable comments on an earlier draft of the manuscript, and The Pew Charitable Trusts for providing support that helped us complete this article.

References cited

- [AR/FE/TU] American Rivers, Friends of the Earth, Trout Unlimited. 1999. *Dam Removal Success Stories: Restoring Rivers through Selective Removal of Dams That Don't Make Sense*. Washington (DC): AR/FE/TU.
- Boardman AE, Greenberg DH, Vining AR, Weimer DL. 1996. *Cost–Benefit Analysis: Concepts and Practice*. Upper Saddle River (NJ): Prentice Hall.
- Dickey E. 1999. Grain transportation after partial removal of the four Lower Snake River dams: An affordable and efficient transportation plan. Washington (DC): American Rivers.
- ECONorthwest. 1999. *An economic strategy for the lower Snake River*. Eugene (OR): ECONorthwest.
- . 2000. *Review of the Draft Lower Snake River Juvenile Salmon Migration FR/EIS*. Eugene (OR): ECONorthwest.
- Hong P, Yang DJ. 1992. Tree-huggers vs. jobs: It's not that simple. *Business Week*, 19 October, pp. 108–109.
- Hughes J. 1999. Gorton calls dam removal nightmare. *Seattle Times*, 28 July, p. B6.
- [NRC] National Research Council. 1999. *New directions in water resources: Planning for the US Army Corps of Engineers*. Washington (DC): National Academy Press.
- Petts GE. 1984. *Impounded Rivers: Perspectives for Ecological Management*. Chichester (UK): John Wiley & Sons.
- Power T. 1996. *Lost Landscapes and Failed Economies: The Search for a Value of Place*. Washington (DC): Island Press.
- Trout Unlimited. 2001. *Small Dam Removal: A Review of Potential Economic Benefits*. Arlington (VA): Trout Unlimited.
- Ulrich R, Ota AK. 1990. A plan that will list steps to protect the owl over its entire range, from British Columbia to San Francisco Bay, is expected Tuesday. *Oregonian*, 23 June, p. A1.
- [USACE] US Army Corps of Engineers. 1999a. *Draft Lower Snake River juvenile salmon migration feasibility report/environmental impact statement*. CD-ROM. Walla Walla (WA): US Army Corps of Engineers, Walla Walla District.
- . 1999b. *Lower Snake River Juvenile Salmon Migration Feasibility Report/Environmental Impact Statement, Appendix I: Economics*. Walla Walla (WA): US Army Corps of Engineers, Walla Walla District.
- [USDC] US Department of Commerce. 1998. *Regional Economic Information System, 1969–1996*. CD-ROM. Washington (DC): USDC, Economics and Statistics Administration, Bureau of Economic Analysis. RCN-0191.
- . 2000. *Regional Economic Information System, 1969–1998*. CD-ROM. Washington (DC): US Department of Commerce, Economic and Statistics Administration, Bureau of Economic Analysis. RCN-0250.
- . 2001. *Regional accounts data*. (9 July 2002; www.bea.doc.gov/bea/regional/reis)
- [USWRC] US Water Resources Council. 1983. *Economic and environmental principles and guidelines for water and related land resources implementation studies*. Washington (DC): US Water Resources Council.
- Warren, D. 1990–2000. *Production, prices, employment, and trade in Northwest forest industries*. Portland (OR): US Department of Agriculture, Forest Service, Pacific Northwest Research Station.
- Whitelaw EW, et al. 1998. Unpublished letter from concerned economists to Governors Kitzhaber, Knowles, Locke, and Wilson, and Premier Clarke regarding the economic issues of salmon recovery.
- Whitelaw EW. 2000. Breaching dam myths. *Oregon Quarterly* 2000 (autumn): 16–20.

The Conceptual Basis for Ecological Responses to Dam Removal

STAN GREGORY, HIRAM LI, AND JUDY LI

Scientists and resource managers have proposed the removal of nonfunctioning dams or dams that cause environmental harm or present unsafe conditions (Poff et al. 1997, Hart and Poff 2002). The basis for assessment of the ecological responses to dam removal and for the design of ecologically effective removal practices is largely conceptual. Particularly in the Pacific Northwest, the adverse effects that large dams have on endangered anadromous salmon require extensive mitigation measures, such as transporting salmon around dams by barge (figure 1), and are a major factor driving dam removal proposals. The introductory article in this series by Hart and Poff (2002) identifies some of the general effects of dams and the responses to dam removal. This article will extend those issues and illustrate the challenges faced in western North America in the removal of high dams, such as the dams on the Elwha and Snake Rivers.

Although more than 75,000 dams have been built in the United States (Shuman 1995), fewer than 500 have been removed. Most dams that have been removed are less than 10 meters (m) high, and no dams higher than 30 m have been removed. Now, however, at least seven high dams in the Pacific Northwest are being reviewed for possible removal (table 1). Citizens and resource managers face a critical question: How much do we know about likely ecological responses to the removal of dams? Stanley and Doyle (2002) have described empirical studies of the ecological responses that follow removal of small dams, and Smith and colleagues (2000) reported on a regional study of those responses in the Pacific Northwest, but no empirical studies of the effects of removing high dams have been conducted.

This article provides a conceptual perspective of the ecological responses to large dam removal, based on our understanding of the structure and function of river ecosystems and on insights gained from small dam removals, where appropriate. We discuss geomorphic responses, hydrologic effects, and several major biological interactions that are affected by

RESOURCE MANAGERS FACE ENORMOUS CHALLENGES IN ASSESSING THE CONSEQUENCES OF REMOVING LARGE DAMS FROM RIVERS AND EVALUATING MANAGEMENT OPTIONS

dams or their removal. These issues are illustrated in the scientific deliberations concerning removal of high dams in two river basins in the Pacific Northwest—the Elwha River and the Lower Snake River.

Geomorphic structure

The fundamental geomorphic change associated with a dam's presence on or removal from a river is the alteration of the longitudinal profile of the river. Dams create a long, flat water surface marked by an abrupt drop in elevation at the dam. After a dam is removed, water levels and channel positions more closely resemble the original morphology of the river, and the sediments that had been stored behind the dam are sculpted by the subsequent river flow. This adjustment to a new longitudinal profile can cause major changes in the distributions of aquatic organisms.

One of the major environmental challenges of removing high dams is the height of sediments behind the reservoir. This

Stanley Gregory (e-mail: Stanley.Gregory@orst.edu) is a professor, and Judy Li is an assistant professor, in the Department of Fisheries and Wildlife at Oregon State University, Corvallis, OR 97331-3803. Hiram Li is a professor and the assistant leader of the Oregon Fisheries Cooperative Unit of the US Geological Survey at Oregon State University. © 2002 American Institute of Biological Sciences.

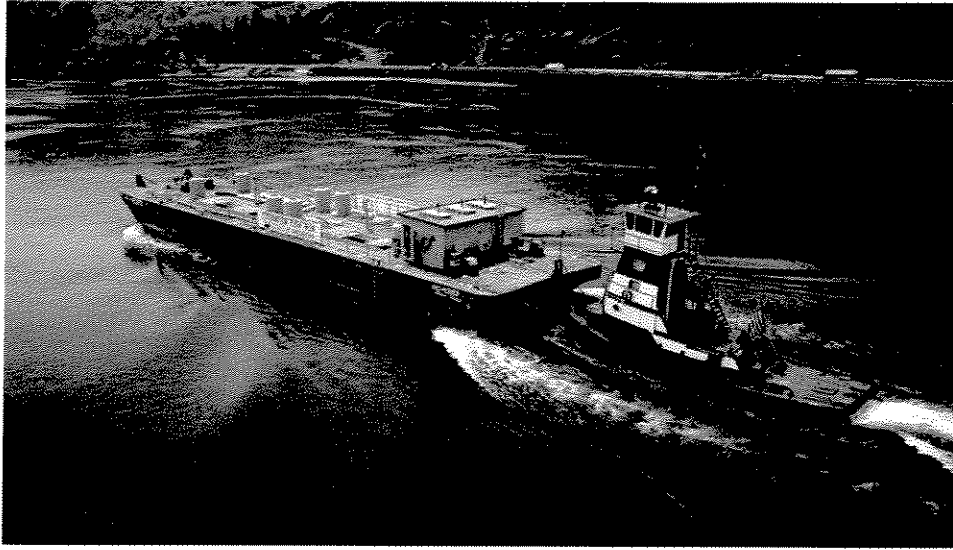


Figure 1. Barge transporting salmon around dams on the Columbia River (photograph from the US Army Corps of Engineers).

is less of a concern with low-head dams or dams in wide valleys, because the vertical relief of the low sediment deposits does not create as much potential for abrupt vertical erosion. The elevation of natural floodplains in most rivers is a small fraction (e.g., less than 1% to 10%) of the width of the bankfull river channel ("bankfull" is a hydrological measure that generally indicates the height or stage of water that just fills the channel). After high dams have been constructed, deposits of sediment upstream of the dam may exceed the relative dimensions of floodplain and bankfull channels found in natural river networks. The removal of a dam with deep sediment deposits may create high, unstable terraces that are accessible to flood waters at the upstream end of the reservoir that existed before the dam's removal but perched far above the channel at the downstream end. The potential for episodic flood erosion of these high terraces and incision of lateral channels into the terraces complicates the restoration of the river and its floodplain after dam removal.

The volume of sediments associated with dams—even low-head dams, in some cases—can have major geomorphic and biological consequences for downstream reaches. Removing a dam can release large volumes of sediment to

downstream reaches over short periods of time and creates easily eroded floodplains. The timing of sediment release and the downstream extent of sediment deposition are difficult to predict, thus leading to a high degree of uncertainty about ecological effects. In addition, subsequent erosion of sediment deposits behind the dam results in frequent and complex channel change within the reach upstream of the dam.

Hydrologic regimes

Dams—particularly hydroelectric and flood-control dams—almost always alter daily, weekly, monthly, and seasonal hydrologic regimes (Poff et al. 1997). Most dams dampen high flows, thereby reducing the beneficial effects of flooding (Junk et al. 1989), such as transporting food into streams from the terrestrial ecosystem, providing floodplain areas for feeding during floods, scouring pools and creating riffles, cleaning silt and fine sediments from gravels, creating deposits of gravel for spawning, and creating complex wood accumulations. In many cases, low flows are augmented to provide water during dry periods and reduce water quality problems caused by point source and non-point source pollutants. As a result, extreme flows, both low and high, are

abbreviated, and their influence in shaping the composition of aquatic communities and ecological processes is greatly reduced. For example, 11 flood control reservoirs were constructed in the Willamette River basin in Oregon from 1948 to 1964. Discharge records from the gaging station at Albany for the period 1893–1997 reveal that low flows (i.e., the daily flow that is exceeded

Table 1. High-head dams in the Pacific Northwest that federal agencies have considered for removal.

Dam	Location	River	Height (meters)	Constructed
Elwha Dam	Washington	Elwha River	32	1914
Glines Dam	Washington	Elwha River	64	1927
Condit Dam	Washington	White Salmon River	36	1913
Ice Harbor Dam	Idaho	Snake River	31	1961
Lower Monumental Dam	Idaho	Snake River	31	1969
Little Goose Dam	Idaho	Snake River	30	1970
Lower Granite Dam	Idaho	Snake River	31	1975

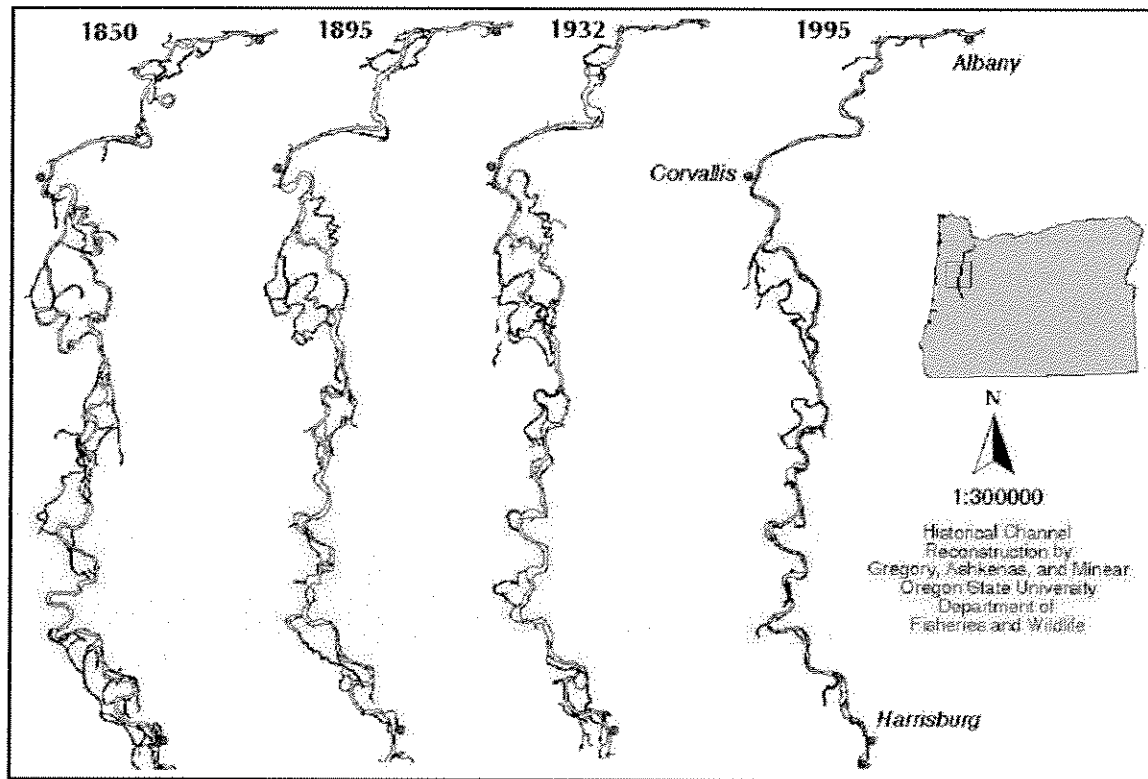


Figure 2. Channels of the Willamette River in 1850, 1895, 1932, and 1995 (from Gregory et al. 2002).

100% of the time) are more than 13 cubic meters per second (m^3/s) greater, and peak flows (i.e., the daily flows that are exceeded less than 0.01% of the time) are almost 2000 m^3/s lower, after dam implementation (table 2).

One of the major ecological benefits of dam removal is the restoration of hydrologic regimes, particularly in the local reach and immediately downstream. Such hydrologic changes are possible through modification of the dam's operation (without its removal), but the benefits to connectivity and geomorphic complexity afforded by removal would not be realized. It is possible to operate a dam to provide low flows and high flows that are similar in magnitude and timing to natural flows, but some modification of flow is inevitable for dams

erected to store and release water to create peaking flow for powering turbines, to dampen flood flows, or to augment low flows. Even if natural flows are closely simulated in dam operation, the geomorphic effects of trapping sediment behind the dam and loss of connectivity for migrating organisms persist.

Interactions of geomorphic and hydrologic processes shape river channels through both erosional and depositional processes that occur during floods that fill the active channel and extend across river floodplains. If large floods are eliminated by dams, channels can incise and impede interaction with their floodplains. In the Willamette River in Oregon, more than 50% of the channel complexity has been reduced through active channel alteration, bank hardening, and hydrologic alteration through flood control (figure 2; Gregory et al. 2002). Though only 26% of the length of riverbanks in the Willamette has been armored by riprap (continuous cover of large boulders) on one or both banks, two-thirds of the meanders in the river are hardened and anchored by riprap and channel dynamics are severely dampened. In general, the more dynamic reaches of river are straightened and altered, and these changes are augmented by lowered peak flows through dam operations. Dam removal potentially restores hydrologic conditions and permits more dynamic channels.

However, a possible unintended consequence is that society may attempt to attain its goals (e.g., flood control, water supply, and commerce) through different means, thereby nullifying gains provided by the new hydrologic and geomorphic conditions created by removal of dams. For

Table 2. Discharge (cubic meters per second) required to meet different exceedance levels in the Willamette River at Albany, Oregon, during two time periods: 1893–1976 and 1969–1997 (data from US Geological Survey flow records).

Percentage exceeded	Flow	
	1893–1976	1969–1997
100	60	73
50	273	267
10	957	933
1	2010	1799
0.1	3398	2533
0.01	5239	3257

example, concerns about uncontrolled flooding after removal of a dam may cause landowners and agencies to attempt to reduce bank erosion through riprap, levees, and other forms of channel hardening. Efforts to “discipline” channels may diminish some of the environmental benefits of dam removal. Similarly, local needs for electrical power may cause communities to turn to other methods of power generation that have other effects on the environment, such as air quality. Dams constructed to provide water supplies may be replaced with water withdrawals from groundwater. Though these alternative actions would not entirely negate the potential benefits of dam removal, decisionmakers are rarely faced with the task of simply removing a dam; the factors that led to its construction continue to influence community actions.

Biological responses

Dams in northwestern rivers influence salmonids and other species by eliminating spawning and rearing habitats in the area covered by reservoirs, changing water velocities that influence migration rates, altering currents that are attractants for migrating fish, forcing some fish through turbines where they experience extreme pressures, increasing river temperatures as the sun warms the slower waters of the reservoir, exposing migrating juvenile fishes to fish and avian predators, and modifying flood patterns that shape river habitats and maintain spawning gravels. Removal of dams potentially restores river temperature patterns, flow patterns for migrating fish, and flood dynamics. The potential negative impacts of dam removal on salmonids are associated primarily with the instabilities of sediments and terraces stored behind the dam. In the case of the Elwha River, planners hope to minimize these effects through temporal phasing of dam removal. In the case of the Snake River dams, federal agencies have examined options that would remove only the earthen portions of the dams and retain the concrete section to stabilize the upstream sediment deposits.

Installation of dams has caused the decline of indigenous aquatic fauna and changes to riparian vegetation worldwide (Li et al. 1987, Pfeiffer and Grace 1987, Friedman and Auble 1999, Hughes and Parmalee 1999, Aparecio et al. 2000, Jansson et al. 2000, Penczak and Kruk 2000, Sharma 2001). Dams influence changes in species diversity in several ways. The stream and riparian habitats are changed by inundation, flow alterations, and influences on groundwater and the water table (Friedman and Auble 1999, Shafroth 1999, Rood and Mahoney 2000). Because dams are barriers that limit the dispersal of organisms and propagules, migration patterns are interrupted, breaking key links in the life history of riverine and aquatic organisms (Andersson et al. 2000, Jansson et al. 2000, Morita et al. 2000).

Direct impacts on survival. In several ways, dams have become killing fields for native aquatic species. Each dam can be thought of as a density-independent source of mortality, a type of predator that kills through the shear forces caused by the cavitation of turbine electrical generators (Coutant and

Whitney 2000). In the Columbia basin, each dam is estimated to kill 5% to 20% of all the juvenile salmonids migrating downstream (Raymond 1979, Skalski 1998). What is not known are the extended effects on survival of salmonids that pass through a number of dams through different migration paths from their natal streams to the ocean, a journey that requires weeks to months for most species of anadromous salmon. Moreover, negotiating each dam causes elevated levels of serum cortisol as a result of stress. This suppresses the immune system and exposes fish to higher risks of disease (Maule et al. 1988). Dams create conditions that cause fishes to die from gas supersaturation, a condition similar to the bends (decompression sickness) in humans (Bouck 1980, Crunkilton and Czarnecki 1980, Penney 1987). When water spills over dams into deep water, atmospheric gases are dissolved in water under high pressure. This can lead to supersaturation of nitrogen at 110%–120% levels (Montgomery and Becker 1980, Ryan et al. 2000). Unless there are shallow areas, such as riffles, where gas levels can equilibrate at the air–water interface, supersaturated conditions can extend for several kilometers. For aquatic organisms that move from deep water up to shallow depths, these conditions can lead to gas bubble disease, in which the supersaturated gases come out of solution in the organisms’ body fluids and cause embolisms. Just as the bends can be fatal to scuba divers who surface too quickly, these gas bubbles can lead to dramatic kills of aquatic organisms.

Indirect effects on nutrients and water quality.

Dams are sediment traps that can keep nutrients such as silica sequestered behind dams, thereby changing community composition of phytoplankton downstream, as witnessed in the Black and Baltic Seas (Humborg et al. 2000). Retention of nutrients behind dams due to the reduced velocity and longer residence time of water in the reach changes the availability of nutrients and composition of plant and microbial communities. Sediment trapping by dams will accumulate and store toxic materials that are adsorbed physically on sediment particles or absorbed actively by the biota attached to the sediments (Dauta et al. 1999). Gravels and cobbles are sequestered behind dams, which limits their recruitment downstream and leads to habitat changes in streams and estuaries (Goselink et al. 1974, Kondolf 1997).

Dams can change the natural variation of stream temperatures, depending upon the dam’s size and mode of operation. Releases of hypolimnetic water (the colder, most dense layer of water in a reservoir that is thermally stratified) from high dams can lower stream temperatures, thereby limiting the reproduction of warmwater fishes and shifting downstream communities to coldwater organisms (Clarkson and Childs 2000). Conversely, low-head dams can act as heat traps and shift community composition in the opposite direction (Walks et al. 2000).

Indirect effects on species interactions. Distributions and abundance of native species can be altered around or

within reservoirs by interactions with either nonnative species or other native species. Dams can create novel habitats, habitats of marginal value to native species, or intensified interactions among species.

Predators. Sediment trapping has clarified normally turbid streams in the Colorado and Missouri basins. One result has been that native fishes are now exposed to greater predation by piscivores (Pfleiger and Grace 1987, Johnson and Hines 1999, Petersen and Ward 1999). Dams in streams of the Columbia basin created migration bottlenecks for migrating salmonids, exposing them to greater contact time with native predators such as northern pikeminnow (*Ptychocheilus oregonensis*) and avian predators (Buchanan et al. 1981). In the American West, native fishes tend to be more adapted to lotic conditions because of the relative scarcity of interconnected streams and lakes. Therefore, lentic fishes or fishes that have evolved in drainages interdispersed with lentic systems, for example, the Upper Mississippi, were introduced (Moyle et al. 1986, Li and Moyle 1999). Gamefishes are almost always carnivorous, and their introduction often foreshadowed a whole suite of novel interactions with the fauna that had not been exposed to their unique traits (Wydowski and Bennett 1981, Li and Moyle 1999). In the Columbia River, the food web has been greatly altered, and the effect of introduced piscivores appears to have increased mortality in an additive fashion (Li et al. 1987, Knutsen and Ward 1999, Ward and Zimmerman 1999). Compensation for this mortality in other stages of the organism's life has not been detected.

The distribution of piscivory varies in different reaches of the Columbia River basin. Predation by introduced sunfish and catfish is higher in the Snake River system, and predation by northern pikeminnow is greater than that by alien piscivores in the lower Columbia River (Zimmerman 1999). In part, this reflects the patchy distribution of habitats and species in the system. In the Columbia River, most of the exotic fishes are located in backwaters and reaches with lentic characteristics, and native fishes are most common in the free-flowing mainstem (Hjort et al. 1981). It also reflects the susceptibility of native fishes to alien predators. As an example, juveniles of fall run chinook salmon (*Oncorhynchus tshawytscha*) are smaller than spring-summer run chinook juveniles, and preference by alien smallmouth bass (*Micropterus dolomieu*) for them may reflect size-selective preferences (Tabor et al. 1993, Zimmerman 1999). As predicted by Li and colleagues (1987), the nocturnal northern sanddollar (*Percopsis transmontana*) appears to be more vulnerable to exotic piscivores, because their size and behavior make them vulnerable to smallmouth bass and the nocturnal walleye (*Stizostedion vitreum*).

In the Colorado River, the combination of the change in seasonal patterns of river discharge, water clarity, temperature, and the introduction of exotic species—all products of regulating the river—complicates recovery of the indigenous minnows and suckers. Radiotracking studies indicate that suitable habitats for native species still exist, but dispersal becomes problematic (Irving and Modde 2000). Although cold

water released from Glen Canyon Dam restricts native fishes in the Little Colorado River from dispersing to downstream reaches, such barriers offer protection from invasive species. The cold water prevents alien warmwater predators (striped bass [*Morone saxatilis*] and largemouth bass [*Micropterus salmoides*]) from dispersing into upstream reaches and invading one of the last strongholds for native fishes in the Colorado River system.

American shad. The American shad (*Alosa sapidissima*) illustrates a paradox that occurs when a species is more abundant in a new area to which it was introduced than in its native range. American shad is a highly prized, native anadromous fish along the Atlantic seaboard of North America. Dams are a primary cause of its severe decline. Many of these dams did not provide fish ladders, thus blocking passage to spawning areas upstream, and altered habitat conditions for pelagic eggs and shad larvae (figure 3a) (Walburg and Nichols 1967). Ironically, this same fish is commonly found spawning in streams along the Pacific Coast, where it is an alien species (Lampman 1946). Introduced to the Sacramento River in 1871, it expanded its range rapidly, and by the 1890s it had reached southeastern Alaska (Welander 1940). To compound the irony, shad population growth exploded exponentially in the Columbia River following the installation of the Dalles Dam in 1960, which inundated Celilo Falls, thereby removing a barrier to upstream movement (figure 3b). Interestingly, this phase coincided with the steep decline of Pacific salmonids and the construction of several high dams.

The paradox can be explained by the fact that the dams, especially in Maine, were barriers, whereas, at least in one instance, the Dalles Dam gave shad access to spawning areas in the upper Columbia River. Further improvements to fish passage facilities may have facilitated expansion of the shad's range to Priest Rapids Dam in the upper mainstem Columbia River and to Lower Granite Dam in the Snake River drainage (Monk et al. 1989). Celilo Falls was not an impediment for Pacific salmonids migrating up the Columbia River; but the mainstem dams reduced salmon migration and also eliminated spawning habitat of fall run chinook salmon, which spawn only in higher-order streams.

A second factor is the effects of commercial harvest on shad populations—at its peak, approximately 50 million pounds were caught off the Atlantic Coast, a figure that is 13 million pounds greater than the highest shad run ever recorded for the Columbia River (from Boschung et al. 1983). Commercial harvest of shad in the Columbia River is small, approximately 740,000 pounds. Populations of shad may continue to expand in the Columbia River, but numbers are still only a fraction of what they must have been historically for shad on the Atlantic Coast. Columbia River fisheries managers, who have noticed the correlation between shad increase and salmon decline, are concerned about the potential for competition for food between shad and salmonids. The alternative explanation for the increase of shad in the Columbia River while Pacific salmon are declining is that the high dams of the Columbia River have opposite effects on shad and salmon.

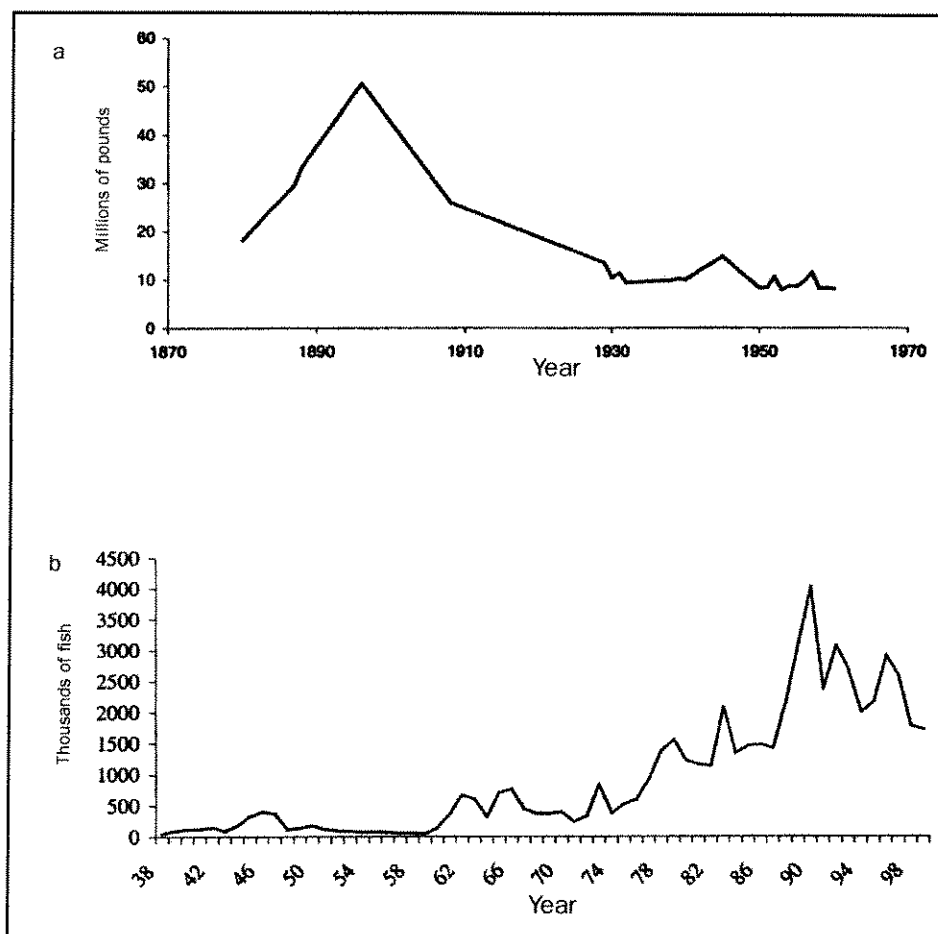


Figure 3. Trends in shad abundances over time. (a) Atlantic Coast shad harvest 1880–1960. (b) Columbia River shad run minimum estimates from 1938–1999.

Until we learn more about shad ecology in the Columbia River and conduct informative experiments and monitoring programs, the factors responsible for declines of shad on the Atlantic Coast and increases in shad on the Pacific Coast, including the influence of dams in the population trends, will remain controversial.

Freshwater mussels. Changes in flow, sediments, and temperatures when dams are removed may have noticeable effects on beds of freshwater bivalves. The hypolimnetic waters that are released by dams prevent gametogenesis and spawning of warmwater mussels (Neves 1999). Fish species upon which mussel species depend during the glochidia stages (when the young mussel larvae are parasitic on fish) may also be affected by alteration of flow and temperatures downstream of dams; loss of host organisms creates an indirect negative effect on mussel colonization and survival. Within a few years of reservoir inundation, water depths and changes in sedimentation eliminate upstream bivalve beds. Because mussels and other bivalves depend on flowing water and unimpeded movements of host fish, dam removal may allow reconnection of populations of bivalves fragmented by lentic waters behind dams.

Case studies

In the Pacific Northwest, dams on the Columbia River system have eliminated access of anadromous salmonids to an estimated 55% of the total area and 33% of the total stream miles (Lichatowich 1999). The National Research Council report on salmon concluded that “as many as 90% of young salmon might survive passage over, around, and through any individual hydropower project on the Columbia–Snake river mainstream” (NRC 1996). For example, fish that must pass through a sequence of five dams with 90% success of passage through each dam would experience a loss of 41% of the original number that attempted to migrate downstream. Such cumulative effects of multiple dams on mainstem rivers are widely accepted as a major influence on the decline of anadromous salmon in the western United States.

The concepts and challenges described above can be illustrated through case studies of specific dams of the Pacific Northwest. The Elwha River dams, which may be the first high-head dams to be removed in the United States, illustrate the possible approaches for

dealing with the challenges created by removing dams that have developed deep sediment deposits upstream (Stoker and Harbor 1991). The Snake River dams provide examples of the complex decisions concerning endangered species and resource uses that removal of major dams entails. Collectively, these case studies point to some of the intricate issues facing decisionmakers in the consideration of the removal of large dams.

Dams of the Elwha River. The Elwha River flows for 72 km north out of the Olympic Mountains to the Straits of Juan de Fuca in northwestern Washington state. It drains more than 161 km of tributary streams and falls 1372 m in elevation, with an average annual instantaneous flow of approximately 43 m³/s (Bureau of Reclamation 1996). More than 80% of the Elwha River basin is located in the Olympic National Park. Elwha Dam was built between 1910 and 1912, 7.9 km upstream of the confluence with the Straits of Juan de Fuca (figure 4). Behind this concrete and earth-fill dam, Lake Aldwell can store 10 million m³ with a surface area of 108 hectares (ha). Up-river 13.7 km, the larger Glines Canyon dam is a tall, 64-m-high, single-arch concrete dam; it holds Lake Mills, which can

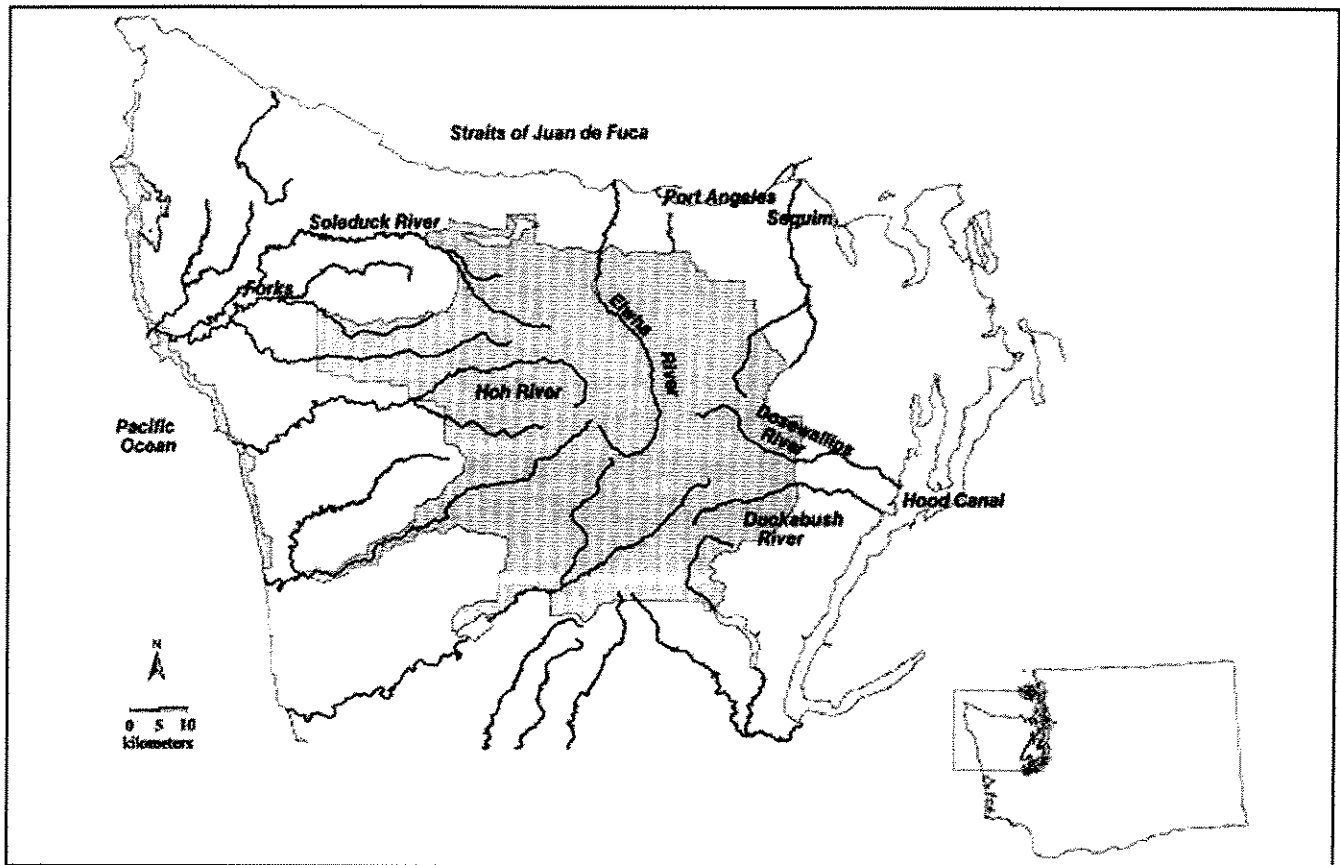


Figure 4. Elwha River on the Olympic Peninsula in Washington (diagram from National Park Service).

contain 50 million m³, with a surface area of 168 ha. Both dams are operated as run-of-river dams (i.e., daily flows are not altered by the dams), but daily hydrological regimes are modified by the dams. But these dams block fish passage and trap more than 13 million m³ of sediment, mostly behind Glines Canyon Dam. Plans to remove these dams to restore this ecosystem, particularly its salmonid runs, received federal approval with the Elwha River Ecosystem and Fisheries Restoration Act (PL 102-495), signed by President Bush in 1992, which authorizes the secretary of the interior to acquire the dams and remove them if their removal is necessary to achieve "the full restoration of the Elwha River ecosystem and native anadromous fisheries." The Department of the Interior purchased the dams in 2000, and they are scheduled to be removed in 2003 if additional funding can be obtained. The size of these dams, the magnitude of river discharge, and volume of sediments behind the dams make the prospective dam removal a much larger undertaking than other projects to date.

The precarious status of salmonids in the Pacific Northwest and the potential gain for spawning habitat of the Elwha's anadromous salmonid stocks provided the impetus for this ambitious project (Wunderlich et al. 1994). There are no precise estimates for historical production in this river, but according to 1987 estimates by the Joint Fish and Wildlife Agencies, there were potentially high numbers of pink (*O. gor-*

buscha) and chum salmon (*O. keta*), and relatively high numbers of chinook and steelhead (*O. mykiss*), compared with other streams in the region. The river was renowned for the size of its fish, especially the chinook, which reportedly weighed more than 100 pounds apiece in the 1930s (FERC 1991). Other salmonids present in the Elwha are anadromous coho (*O. kisutch*), resident and anadromous Dolly Varden (*Salvelinus malma*), and sea-run cutthroat (*O. clarki*). Hatchery coho, steelhead, and chinook are important competitors in the lower reaches. Pink salmon runs have declined rapidly since 1979, plummeting from runs of about 40,000 in 1959 to mere hundreds in recent years. Reduction of pink and chum salmon was related to predation by hatchery fish in other systems (Johnson 1973, Cardwell and Fresh 1979), and predation may be an important factor in the lower Elwha as well.

The uppermost river reach is very steep, and most of the main river to Lake Mills meanders through alluvial deposits, sometimes flowing through steep canyons. A major waterfall at 55 km upstream from the mouth would limit some species from using much of the upper river reach. At present the middle reach is highly armored and dominated by cobble and large boulders. The short-term effects of dam removal will include the redistribution of large volumes of silt downstream (Stoker and Harbor 1991), but eventually additions of gravels will open up extensive reaches of usable spawning habi-

tat in the middle reach. These changes would likely affect species that occupy the lower reaches of the river the most (e.g., chinook, pink, and chum salmon).

Return of anadromy could also affect food webs upstream. For example, resident steelhead and Dolly Varden would lose some spawning habitats associated with reservoirs and also be subject to greater competition and predation by juveniles of other salmonid species. Increased fish densities will benefit piscivorous predators such as common mergansers, great blue heron, and belted kingfishers. Presently, bald eagles, whose numbers are highly correlated to chum escapement in other Olympic watersheds, are uncommon (only six observations in winter 1990). Their scarcity is likely due to lack of prey. Based on estimates from the nearby Skagit River where bald eagles are numerous, 18,000 chum would attract about 140 bald eagles to the drainage (DellaSala et al. 1990). Availability of anadromous salmonids as prey for bald eagles will depend on coincidence of fish migrations and eagle arrivals. Salmon with migratory patterns less synchronized than chum with eagle movements, particularly pinks and chinook, could provide food only for eagles arriving in early winter.

The lake-like conditions of the reservoir reaches have created favorable conditions for almost a century for some plants and animals that will be adversely affected by dam removals. Shoreline cover along Lake Aldwell will greatly diminish and thus significant habitat for lacustrine mink will be removed (FERC 1991). Surprisingly, beaver are likely to increase with recolonization of hardwoods along riverine terraces. Wetland biomes that have developed along lake edges will disappear with their associated plants, one of them a bicolored linanthus unique to the Elwha valley (FERC 1991). Eventually other wetlands are expected to develop along stabilized backwater and meanders of the reestablished floodplains.

Overall, removing the dams will greatly enhance anadromous fish runs and, consequently, food chains. Dramatic increases in salmon carcasses are expected to provide nutrients and food resources to juvenile fishes and other aquatic predators. Changes in hydrology and return to natural flow patterns will influence downstream temperatures and instream dynamics. Average temperatures in the middle and lower river reaches will be lower than at present. Maximum daily water temperatures are 15°C–20°C in low water years (Washington Department of Fisheries, Elwha hatchery records); these levels are most likely harmful to fish eggs. These elevated water temperatures may increase the infection rate of *Dermocystidium* bacteria, which attack salmonids as they come from marine systems into fresh water (FERC 1991). Lowered water temperatures after dam removal would decrease the incidence of the disease and thus potentially increase salmonid survival.

In 1990, invertebrates that were collected downstream of Glines Canyon Dam were significantly less diverse than in upstream reaches; they were characterized by early-colonizing species and abundant filter-feeding caddis flies. Dominance of baetid mayflies and chironomid midges reflected the almost

daily fluctuating flows as dam releases pulsed through the system. Though the dams were managed as run-of-the-river flows, daily fluctuations were not conducive for stable invertebrate populations, especially organisms in habitats associated with the river margin, with potentially similar effects on young fish. More naturally predictable flows will contribute to increases in productivity at all levels.

Where the Elwha River flows into the Straits of Juan de Fuca, its estuary supports clam beds. When the Elwha and Glines Canyon dams are removed, an estimated increase of 160,000 m³ in sediments will be supplied at the mouth of the river (FERC 1991). It is possible that these sediments will have short-term impacts on downstream communities and nearshore marine benthic communities and shellfish at the mouth of the Elwha River. During the years of the dams' operation, there has been a dramatic reduction of sediments (from approximately 115,000 to only 1835 m³ per year). Coastal sediments have been reduced by 36%. The sand, gravel, and cobbles that would be reintroduced into the coastal zone may provide sufficient sediments to support a small increase in shellfish.

Dams of the Snake River. The Snake River, which once produced 45% of all chinook salmon found in the Columbia River basin (Hassemer et al. 1997), has four dams—Ice Harbor, Lower Monumental, Little Goose, and Lower Granite Dams—that affect species of anadromous salmon listed as endangered under the Endangered Species Act (figure 5). A study of historical patterns of survival of different stocks of chinook salmon in the Columbia River basin concluded that survival dropped sharply in reaches affected by dams soon after construction, but survival did not change abruptly in reaches not influenced by dam construction (Schaller et al. 1999). Removal of these dams might decrease the risk of extinction for these species. Coho salmon are now extinct in the Snake River. Sockeye were listed as endangered in 1992, and spring chinook, summer chinook, fall chinook, and steelhead were listed as threatened from 1993 to 1998. In February 2001, federal courts ruled that the US Army Corps of Engineers was required to comply with the Clean Water Act in its management of dams. The courts determined that the dams caused temperature increases and gas supersaturation that exceeded limits under the Clean Water Act. Future dam management operations must address the water quality goals and policies of the state and federal governments. Most management actions have focused on reducing effects of the dam by retrofitting dams with better passage facilities, trucking and barging the fish around the dams, or increasing the spill of water over the dams so that fewer fish went through the turbines. But all of these actions essentially coexist with the existing dams. The economic impacts have been hotly debated (see Whitelaw and MacMullan 2002), but the consequences of dam removal on the risk of extinction of salmonids also has been controversial. More than 200 scientists signed a letter to President Clinton calling for removal of the four Snake River dams.

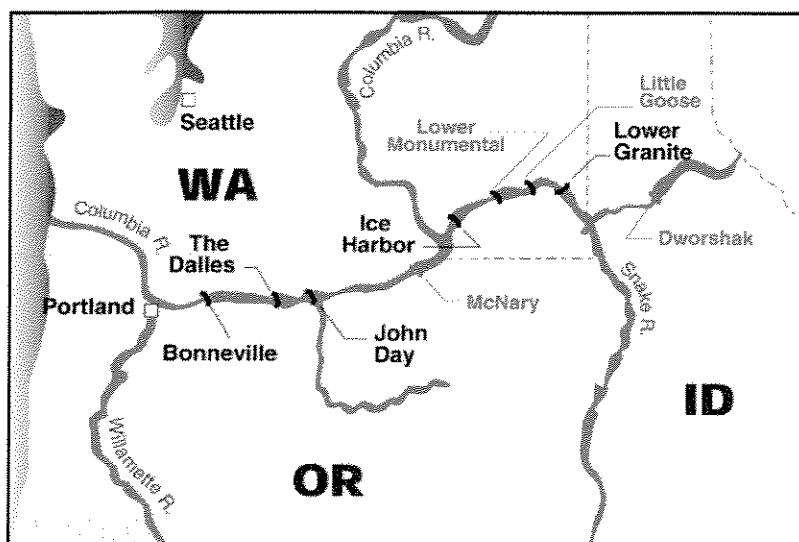


Figure 5. Dams on the Columbia River. The four Snake River dams considered for removal are on the Snake River immediately upstream of the confluence with Columbia River (diagram from US Army Corps of Engineers).

Recently, the US National Marine Fisheries Service estimated that anadromous salmon stocks have a 55%–100% probability of extinction over the next 100 years (NMFS 2000). Despite dam passage improvements that have dramatically mitigated direct mortality associated with dams, the NMFS concluded that the removal of the dams would not reduce the risk of extinction under current conditions. They found that Snake River spring–summer chinook salmon would probably continue to decline toward extinction, and therefore NMFS recommended “modest reductions in first-year mortality or estuarine mortality [to] reverse current population declines” (Kareiva et al. 2000). Other scientists modeled these populations incorporating delayed first-year mortality. They considered the possibility that juvenile fish migrating downstream to the ocean might experience delayed mortality. If fish die later because of stress or injury, simple estimates of fish mortality as they pass directly through the immediate vicinity of a dam may be substantially lower than actual mortality. Modeling runs that incorporate higher delayed mortality rates indicated that removal of the Snake River dams could potentially reverse declines in Snake River chinook salmon (Dambacher et al. 2001). Recent analysis of several data sources and modeling concluded that salmon smolts migrating through the dams experience delayed mortality (Budy et al. 2002). All studies (Marmorek and Peters 1998, NMFS 2000, Budy 2001) reviewed by these authors concluded that fish that migrated through the hydrosystems in the river had survival rates that were approximately 25% to 50% lower than those for fish that were transported around the entire hydrosystem.

Studies suggest that upriver salmon will not benefit from the breaching of the Snake River dams (Kareiva et al. 2000, Zabel and Williams 2000), but other interpretations of the data

differ if delayed mortality is considered (Marmorek and Peters 1998, Nemeth and Keifer 1999, Schaller et al. 1999, 2000, Dambacher et al. 2001, Petrosky et al. 2001). Questions about mortality rates of different life history stages of anadromous salmon point to the need for better information about the impacts of human actions on salmonid populations and life history stages.

Connecting science and policy

Ecological responses to dam removal cannot be predicted with a high degree of certainty in complex river ecosystems. Public values and social actions also have large effects on ecosystems and the nature of resource decisions. Most people are reluctant to deconstruct anything they built and financed, even if they later realize that the decision may have been flawed. Resource managers must make critical decisions in the face of uncertainty and complicated social values. In such cases, one approach is the application of a precautionary philosophy in conjunction with the concepts

of adaptive management. The precautionary principle generally suggests that, in the face of uncertainty, efforts to reduce impacts are prudent and reversible choices should be favored over irreversible choices (Ludwig et al. 1993). But this too may be inadequate unless we integrate the larger cultural backdrop (social, economic, political, and legal aspects) concerning decisionmaking. Blumm and colleagues (1998) suggest that they have made this complete analysis for the Snake River dams. In their judgment, breaching these is the most logical step. Additional analysis will improve the basis for this decision, but the technical data will always be limited and decisionmakers will be forced to consider the weight of evidence and will have to make very difficult social and environmental decisions.

Dam removal or the breaching of dams will be controversial in many cases because of the many vested social and political interests. The role of science in forming policy is rapidly changing, and public confusion over the positions of dueling scientists is not uncommon. The current debate surrounding the Snake River dams and the dams of the Elwha River in Washington illustrates the high degree of uncertainty inherent in projecting ecological responses to dam removal. The first challenge is the complexity of physical responses in the naturally variable environments of river systems. Projections of geomorphic and hydrologic changes are not simple and will vary greatly based on local landscapes and climate. Ecological interactions are complex because of the interactions between adjacent terrestrial and aquatic ecosystems, predator–prey interactions, competition, succession, and dispersal of aquatic and terrestrial organisms. Even more complex is the array of social actions in river systems that dictate ecological responses, such as hydrologic alteration, water diversion, bank hardening, land use conversion, exotic species

introductions, and water quality impairment. Resource managers and the public must recognize that precise predictions of ecological change after dam removal are not possible. Nevertheless, the conceptual framework provided by our knowledge of stream ecosystems and their interactions within the landscape provide a basis for prudent choices and adaptive management to local responses to dam removal.

Acknowledgments

Publication of this paper was supported, in part, by the Thomas G. Scott Publication Fund, Oregon State University. Figure 3 was prepared by Peter Barber, Josh Hanson, and Steve Starceovich.

References cited

- Andersson E, Nilsson C, Johansson ME. 2000. Effects of river fragmentation on plant dispersal and riparian flora. *Regulated Rivers: Research and Management* 16: 83–89.
- Aparicio E, Vargas MJ, Olmo JM, de Sostoa A. 2000. Decline of native freshwater fishes in a Mediterranean watershed on the Iberian Peninsula: A quantitative assessment. *Environmental Biology of Fishes* 59: 11–19.
- Blumm MC, Lucas LJ, Miller DB, Rohlf DJ, Spain GH. 1998. Saving Snake River water and salmon simultaneously: The biological, economic, and legal case for breaching the Lower Snake River dams, lowering John Day Reservoir, and restoring natural river flows. *Environmental Law* 28: 997–1054.
- Boschung HT Jr, Williams JD, Gotshall DW, Caldwell DK, Caldwell MC. 1983. *The Audubon Society Field Guide to North American Fishes, Whales and Dolphins*. New York: Alfred A. Knopf.
- Bouck GR. 1980. Etiology of gas bubble disease. *Transactions of the American Fisheries Society* 109: 703–707.
- Buchanan DV, Hooton RM, Moring JR. 1981. Northern squawfish (*Ptychocheilus oregonensis*) predation on juvenile salmonids in sections of the Willamette River basin, Oregon. *Canadian Journal of Fisheries and Aquatic Sciences* 38: 360–364.
- Budy P. 2001. Analytical approaches to assessing recovery options for Snake River chinook salmon. *Utah Cooperative Fish and Wildlife Research Unit* 2001 (1): 1–86.
- Budy P, Theide GP, Bouwes N, Petrosky CE, Schaller H. 2002. Evidence linking delayed mortality of Snake River salmon to their earlier hydrosystem experience. *North American Journal of Fisheries Management* 22: 35–51.
- Bureau of Reclamation. 1996. *Plan for Removal of Elwha and Glines Canyon Dams*. Boise (ID): US Department of Interior, Bureau of Reclamation. Elwha Technical Series no. PN-95-7.
- Cardwell RD, Fresh KL. 1979. Predation upon juvenile salmon. Olympia (WA): Washington Department of Fish and Wildlife.
- Clarkson RW, Childs MR. 2000. Temperature effects of hypolimnial-release dams on early life stages of Colorado River basin big-river fishes. *Copeia* 2000: 402–412.
- Coutant CC, Whitney RR. 2000. Fish behavior in relation to passage through hydropower turbines: A review. *Transactions of the American Fisheries Society* 129: 351–380.
- Crunkilton RL, Czarnecki JM. 1980. Severe gas bubble disease in a warmwater fishery in the midwestern United States. *Transactions of the American Fisheries Society* 109: 725–733.
- Dambacher JM, Rossignol PA, Li HW, Emlen JM. 2001. Dam breaching and chinook salmon recovery. *Science* 291: 939.
- Daut A, Lapaquellerie Y, Maillat N. 1999. Role of the dams on the River Lot on two types of pollution: Point-sources (cadmium) and non-point sources (phosphorus). *Hydrobiologia* 410: 307–314.
- DellaSala D, McShane MC, Stalmaster MV, Mills L, Taylor R. 1990. Bald eagle abundance and chum salmon escapement on the Skagit River, 1978–1989. Paper presented at the annual meeting of the Wildlife Society, Washington and Oregon chapters; 1–2 March 1990; Mt. Hood, OR.
- [FERC] Federal Energy Regulatory Commission, Office of Hydropower Licensing. 1991. *Draft Environmental Impact Statement, Glines Canyon (FERC No. 588) and Elwha (FERC No. 2683) Hydroelectric Projects*, Washington. Washington (DC): Office of Hydropower Licensing. Report no. 94-18970.
- Friedman JM, Auble GT. 1999. Mortality of riparian box elder from sediment mobilization and extended inundation. *Regulated Rivers: Research and Management* 15: 463–476.
- Gosselink JG, Odum EP, Pope RM. 1974. The value of the tidal marsh. Baton Rouge: Louisiana State University, Center for Wetland Resources. Publication no. LSU-SG-74-03.
- Gregory SV, Ashkenas LR, Oetter D, Minear P, Wildman K. 2002. Historical Willamette River channel change. Pages 18–25 in Hulse D, Gregory S, Baker J, eds. *Willamette River Basin Atlas: Trajectories of Environmental and Ecological Change*. Corvallis: Oregon State University Press.
- Hart DD, Poff NL. 2002. A special section on dam removal and river restoration. *BioScience* 52: 653–655.
- Hassemer PF, Kiefer SW, Petrosky CE. 1997. Idaho's salmon: Can we count every last one? Pages 113–125 in Stouder DG, Bisson PA, Naiman RJ, eds. *Pacific Salmon and Their Ecosystems: Status and Future Options*. London: Chapman and Hall.
- Hjort RC, Mundy BC, Hulett PL. 1981. *Habitat requirements for resident fishes in the reservoirs of the lower Columbia River*. Final Report to US Army Corps of Engineers, Portland, OR. Corvallis: Oregon State University, Department of Fisheries and Wildlife. Contract no. DACW57-79-C-0067.
- Hughes MH, Parmalee PW. 1999. Prehistoric and modern freshwater mussel (Mollusca: Bivalvia: Unionidae) faunas of the Tennessee river: Alabama, Kentucky, and Tennessee. *Regulated Rivers: Research and Management* 15: 25–42.
- Humborg C, Conley DJ, Rahm L, Wulff F, Cociasu A, Ittekkot V. 2000. Silicon retention in river basins: Far-reaching effects on biogeochemistry and aquatic food webs in coastal marine environments. *Ambio* 29: 45–50.
- Irving DB, Modde T. 2000. Home-range fidelity and use of historic habitat by adult Colorado pikeminnow (*Ptychocheilus lucius*) in the White river, Colorado and Utah. *Western North American Naturalist* 60: 16–25.
- Jansson R, Nilson C, Dynesius M, Andersson E. 2000. Effects of river regulation on river-margin vegetation: A comparison of eight boreal rivers. *Ecological Applications* 10: 203–224.
- Johnson JE, Hines RT. 1999. Effect of suspended sediment on vulnerability of young razorback suckers to predation. *Transactions of the American Fisheries Society* 128: 648–655.
- Johnson RC. 1973. Potential interspecific problems between hatchery coho smolt and juvenile pink and chum salmon. Olympia (WA): Washington Department of Fish and Wildlife.
- Junk WJ, Bayley PB, Sparks RE. 1989. The flood pulse concept in river-floodplain systems. Pages 110–127 in Dodge DP, ed. *Proceedings of the International Large River Symposium*. Ottawa, Ontario (Canada): Department of Fisheries and Oceans. Canadian Special Publication of Fisheries and Aquatic Sciences 106.
- Kareiva P, Marvier M, McClure M. 2000. Recovery and management options for spring/summer chinook salmon in the Columbia River basin. *Science* 290: 977–979.
- Knutsen CJ, Ward DL. 1999. Biological characteristics of northern pikeminnow in the Lower Columbia and Snake Rivers before and after sustained exploitation. *Transactions of the American Fisheries Society* 128: 1008–1020.
- Kondolf GM. 1997. Hungry water: Effects of dams and gravel mining on river channels. *Environmental Management* 21: 533–551.
- Lampman BH. 1946. *The Coming of the Pond Fishes*. Portland (OR): Benford and Mart.
- Li HW, Moyle PB. 1999. Management of exotics. Pages 345–374 in Kohler C, Hubert W, eds. *Inland Fisheries Management in North America*. 2nd ed. Bethesda (MD): American Fisheries Society.
- Li HW, Schreck CB, Bond CE, Rexstad E. 1987. Factors influencing changes in fish assemblages of Pacific Northwest streams. Pages 193–202 in Matthews WJ, Heins DC, eds. *Community and Evolutionary Ecology of North American Stream Fishes*. Norman: University of Oklahoma Press.
- Lichtowich J. 1999. *Salmon without Rivers*. Washington (DC): Island Press.

- Ludwig D, Hilborn R, Walters C. 1993. Uncertainty, resource exploitation, and conservation: Lessons from history. *Science* 260: 35–36.
- Marmorek DR, Peters CN, eds. 1998. Plan for Analyzing and Testing Hypotheses (PATH): Final Report for Fiscal Year 1998. Vancouver (Canada): ESSA Technologies.
- Maule AG, Schreck CB, Bradford CS, Barton BA. 1988. Physiological effects of collecting and transporting emigrating juvenile chinook salmon past dams on the Columbia River. *Transactions of the American Fisheries Society* 117: 245–261.
- Monk BD, Weaver D, Thompson C, Ossiander F. 1989. Effects of flow and weir design on the passage behavior of American shad and salmonids in an experimental fish ladder. *North American Journal of Fisheries Management* 9: 60–67.
- Montgomery JC, Becker CD. 1980. Gas bubble disease in smallmouth bass and northern squawfish from the Snake and Columbia Rivers. *Transactions of the American Fisheries Society* 109: 734–736.
- Morita K, Yamamoto S, Hoshino N. 2000. Extreme life history change of white-spotted char (*Salvelinus leucomaenis*) after damming. *Canadian Journal of Fisheries and Aquatic Sciences* 57: 1300–1306.
- Moyle PB, Li HW, Barton BA. 1986. The Frankenstein effect: Impact of introduced fishes on native fishes in North America. Pages 415–426 in Stroud RH, ed. *Fish Culture in Fisheries Management*. Bethesda (MD): American Fisheries Society, Fish Culture Section and Fisheries Management Section.
- [NRC] National Research Council. 1996. *Upstream: Salmon and Society in the Pacific Northwest*. Washington (DC): National Academy Press.
- Nemeth DJ, Kiefer RB. 1999. Snake River spring and summer chinook salmon—the choice for recovery. *Fisheries* 24: 16–22.
- Neves RJ. 1999. Be dammed: The life and times of freshwater mussel populations in the United States. Paper presented at the annual meeting of the American Fisheries Society; 1 September 1999; Charlotte, NC.
- [NMFS] US National Marine Fisheries Service. 2000. Biological opinion of the Federal Columbia River Power System, Including the Juvenile Fish Transportation Program and the US Bureau of Reclamation's 31 dams, Including the Entire Columbia Basin Project. Seattle (WA): NMFS.
- Penczak T, Kruk A. 2000. Threatened obligatory riverine fishes in human-modified Polish rivers. *Ecology of Freshwater Fishes* 9: 109–117.
- Penney GH. 1987. Dissolved oxygen and nitrogen concentrations in Macataqua area waters, 1968, 1969, and 1972. *Canadian Manuscript Report of Fisheries and Aquatic Sciences* 1906: 1–22.
- Petersen JH, Ward DL. 1999. Development and corroboration of a bioenergetics model for northern pikeminnow feeding on juvenile salmonids in the Columbia River. *Transactions of the American Fisheries Society* 128: 784–801.
- Petrosky CE, Schaller HA, Budy P. 2001. Productivity and survival rate trends in the freshwater spawning and rearing stage of Snake River chinook salmon (*Oncorhynchus tshawytscha*). *Canadian Journal of Fisheries and Aquatic Sciences* 58: 1196–1207.
- Pfleger WL, Grace TB. 1987. Changes in the fish fauna of the lower Missouri river, 1940–1983. Pages 155–165 in Matthews WJ, Heins DC, eds. *Community and Evolutionary Ecology of North American Stream Fishes*. Norman: University of Oklahoma Press.
- Poff NL, Allan JD, Bain MB, Karr JR, Prestegard KL, Richter BD, Sparks RE, Stromberg JC. 1997. The natural flow regime. *BioScience* 47: 769–784.
- Pohl MM. 2000. The dams of the Elwha River, Washington: Downstream impacts and policy implications. *Dissertation Abstracts International Part B: Science and Engineering* 60: 3174.
- Raymond HL. 1979. Effects of dams and impoundments on migrations of juvenile chinook salmon and steelhead from the Snake River, 1966 to 1975. *Transactions of the American Fisheries Society* 108: 505–529.
- Rood SB, Mahoney JM. 2000. Revised instream flow regulation enables cottonwood recruitment along the St. Mary River, Alberta, Canada. *Rivers* 7: 109–125.
- Ryan BA, Dawley EM, Nelson RA. 2000. Modeling the effects of supersaturated dissolved gas on resident aquatic biota in the main-stem Snake and Columbia Rivers. *North American Journal of Fisheries Management* 20: 192–204.
- Schaller HA, Petrosky CE, Langness OP. 1999. Contrasting patterns of productivity and survival rates for stream-type chinook salmon (*Oncorhynchus tshawytscha*) populations of the Snake and Columbia Rivers. *Canadian Journal of Fisheries and Aquatic Sciences* 56: 1031–1045.
- . 2000. Reply to Zabel and Williams' comments on "Contrasting patterns of productivity and survival rates for stream-type chinook salmon (*Oncorhynchus tshawytscha*) populations of the Snake and Columbia Rivers." *Canadian Journal of Fisheries and Aquatic Sciences* 57: 1742–1746.
- Shafroth P. 1999. Downstream effects of dams on riparian vegetation: Bill Williams river, Arizona. *Dissertation Abstracts International, Part B: Science and Engineering* 60: 928.
- Sharma VK. 2001. Post impoundment changes in river Beas after Pong Dam emergence. *Applied Fisheries and Aquaculture* 1: 9–12.
- Shuman JR. 1995. Environmental considerations for assessing dam removal alternatives for river restoration. *Regulated Rivers: Research and Management* 11: 295–261.
- Skalski JR. 1998. Estimating season-wide survival rates of outmigrating salmon smolt in the Snake River, Washington. *Canadian Journal of Fisheries and Aquatic Sciences* 55: 761–769.
- Smith LW, Dittmer E, Prevost M, Burt DR. 2000. Breaching of a small irrigation dam in Oregon: A case history. *North American Journal of Fisheries Management* 20: 205–219.
- Stanley EH, Doyle MW. 2002. A geomorphic perspective on nutrient retention following dam removal. *BioScience* 52: 693–701.
- Stoker B, Harbor J. 1991. Dam removal methods, Elwha river, Washington. Pages 668–673 in Shane RM, ed. *Hydraulic Engineering: Proceedings of the 1991 National Conference of the American Society of Civil Engineers*; 29 July–2 August; Nashville, TN. New York: ASCE.
- Tabor RA, Shively RS, Poe TP. 1993. Predation on juvenile salmonids by smallmouth bass and northern squawfish in the Columbia River near Richland, Washington. *North American Journal of Fisheries Management* 13: 831–838.
- Walburg CH, Nichols PR. 1967. Biology and management of the American shad and status of the fisheries, Atlantic Coast of the United States, 1960. Washington (DC): US Department of Interior, Fish and Wildlife Service. *Special Scientific Report—Fisheries no. 550*.
- Walks DJ, Li HW, Reeves GH. 2000. Trout, summer flows, and irrigation canals: A study of habitat condition and trout populations within a complex system. Pages 115–125 in Cowx IG, ed. *Management and Ecology of River Fisheries*. London: Blackwell Science.
- Ward DL, Zimmerman MP. 1999. Response of smallmouth bass to sustained removals of northern pikeminnow in the lower Columbia and Snake Rivers. *Transactions of the American Fisheries Society* 128: 1020–1035.
- Welander AD. 1940. Notes on the dissemination of shad, *Alosa sapidissima* (Wilson), along the Pacific Coast of North America. *Copeia* 1940: 221–223.
- Whitelaw E, MacMullan E. 2002. A framework for estimating the costs and benefits of dam removal. *BioScience* 52: 724–730.
- Wunderlich RC, Winter BD, Meyer JH. 1994. Restoration of the Elwha River ecosystem. *Fisheries* 19(8): 11–19.
- Wydowski RS, Bennett DH. 1981. Forage species in lakes and reservoirs of the western United States. *Transactions of the American Fisheries Society* 110: 764–772.
- Zabel RW, Williams JG. 2000. Comments on "Contrasting patterns of productivity and survival rates for stream-type chinook salmon (*Oncorhynchus tshawytscha*) populations of the Snake and Columbia Rivers." *Canadian Journal of Fisheries and Aquatic Sciences* 57: 1739–1741.
- Zimmerman MP. 1999. Food habits of smallmouth bass, walleyes and northern pikeminnow in the lower Columbia River basin during outmigration of juvenile anadromous salmonids. *Transactions of the American Fisheries Society* 128: 1036–1054.

What Goes Up, May Come Down

BRUCE BABBITT

Once upon a time (not long ago, during the early years of my tenure at the US Department of the Interior), vigilant watershed-based communities could often be seen gathered on the banks of their streams, waving homemade signs, heckling politicians, and vowing to lie down in front of government bulldozers, united by their singular passion to Stop Dams! Save Our River!

No longer.

Rapid developments in the last few years have given rise to a new protest movement. This one, also grassroots, musters its forces, raises funds, organizes committees, shouts down federal officials, lobbies legislators, waves placards for the press, and promises civil disobedience if the government proceeds with its engineering plans. Only this time, the words painted on their banners read "Stop Rivers! Save Our Dam!"

Yet this is only the most visible level of irony. Peel away simple media images and a second level emerges: Ideological opposites on dams are now reversing roles, exchanging hats, and switching arguments with each other.

Consider: Dam opponents once were the true fiscal conservatives. They urged caution, pointing out that dams typically cost taxpayers more than estimated in early projections, and that alternatives to dams exist. They favored the status quo, noting that there was too much uncertainty to allow concrete pouring without more and more laborious environmental and social impact studies. They championed the rights of human lives and livelihoods for Native American tribes who earned their food and made their homes on the banks of rivers but who were still displaced involuntarily and without meaningful compensation.

Back then, it was the dam builders who were pressing for rapid change. They said, "Trust us"—with all the benefits their dam would surely provide, all other concerns would be sorted out...once the dam was under way. They just instinctively *felt* that construction was the right way. Dams were the panacea for floods or fire, irrigation or navigation, voltage or storage. Dams, we were confidently assured, were the answer.

And so they seemed, at the time, to many.

LEARNING FROM OUR EXPERIENCES WITH DAM CONSTRUCTION IN THE PAST CAN GUIDE AND IMPROVE DAM REMOVAL IN THE FUTURE

But gradually, over the years, water evaporated from reservoir surfaces or got choked by algal blooms; concrete crumbled under pressure and time; structures severed salmon migration, collected silt, and cost millions to repair or replace. Scientific studies of unforeseen negative impacts mounted. Slowly, then quickly, dam removal became an answer as well. It became a means for restoring ecologically degraded rivers.

Now, the pro-dam lobby is the one making the conservative case for fiscal austerity, blasting some removals as too expensive, arguing that local owners, ratepayers, and taxpayers (who benefited from a dam) should not have to finance the dam's deconstruction (even though in many cases dam removal is the least expensive option in dam decisions). They demand economic compensation for any displaced downstream irrigators, sawmill operators, energy consumers, or marina owners (who in turn once displaced the tribal fishermen). They agitate over the social rights of landowners around the reservoir, who moved in thinking the dam would stand safely forever, cost free. Surely, they say, other options must be pursued. Most ironic, it is the pro-dam lobby that presses for extensive, time-consuming environmental studies about potential impacts of removal in the face of uncertainty.

It would seem easy to brush off such concerns as hypocritical, given the dam proponents' earlier blasé expediency or their rush to press new dams elsewhere. With dams on the

Bruce Babbitt, former secretary of the Department of the Interior, is an attorney with Latham & Watkins, 555 11th Street, NW, Suite 1000, Washington, DC 20004-1304. © 2002 American Institute of Biological Sciences.

defensive, it is tempting to say, "Trust us—removal is the answer." For quite often, removal does make the most ecological and economic sense.

And lest there be any misunderstanding, my own stand on consensus-based dam removal is on the record. It became increasingly pronounced over the past half-decade as I graduated from one level to the next, embracing sledgehammer, jackhammer, wrecking ball, sky crane, and even C-4 plastic explosives to help dismantle dozens of obsolete structures, structures that had either outlived their function or outweighed their benefits with costs that society was no longer willing to pay.

The change has come. The heyday of dams has come and gone. From my perspective, there is no turning back.

Yet some questions over dam removal linger and should not be too quickly or easily dismissed. They deserve thoughtful answers and, more important, scientific follow-up documentation to back those answers up. The impacts of dam de-construction should be carefully estimated before removal and objectively evaluated afterward, even if—especially if—both predam and postdam examinations were never conducted when the dam was built. And dam proponents should be recognized even by—especially by—those same dam opponents who were excluded from past decisions to build.

Why? Why hold dam removal to a higher standard than construction ever faced? Because if such concerns go unanswered, the future of dam removal may eventually erode to become as vulnerable, unstable, and obsolete as some of the dams it will erase.

In one respect, the dam removal juggernaut is proceeding on solid ground. As shown in this issue of *BioScience*, many leaders at the local and national levels follow the precautionary principle, asking the right questions and raising issues in advance. Moreover, in-depth explorations that address broad stakeholder interests are being conducted by the Heinz Center and Aspen Institute, among others. Some dams are better candidates for removal than others, such as those where the benefits of removal outweigh the uses and benefits of the dam. And scientific study can help to identify the best candidates.

Rather than simply exchange the old simple approach to dams (build now, ask questions later) with a new, equally simple plan (remove now, analyze outcomes later), these initiatives have begun to recognize the socioeconomic and ecological complexity of what we are doing, and they affirm our obligation to the past, to each other, and to our surroundings. In carrying out our obligation, we can use what we have learned from the impacts of dams to help model, predict, and monitor the impacts of their absence.

Science has made it increasingly and painfully clear that a single dam can produce impacts that extend the entire length

of a river and beyond, damaging nearby estuaries, beaches, and ocean and adversely affecting biodiversity on a regional scale. Likewise, we must continue to use science to inform and explain the costs and benefits of removal throughout the watershed.

This can take time. Edwards Dam on the Kennebec River in Maine underwent years of study before its removal. At Savage Rapids Dam, another prime removal candidate on the Rogue River watershed, environmental and economic impact studies go back more than a decade.

It can also involve watershed economies. Before undertaking removal of Glines Canyon and Elwha Dams on the Olympic

Peninsula of Washington, the Department of the Interior compiled a history of impacts on fisheries in the watershed and began modeling the expected impacts of silt changes in the river and at the mouth of the stream to ensure that the final decision incorporated shellfish harvests at the delta as much as angling revenues in the headwaters.

In addition, estimating impacts in advance can save time and money. The Bureau of Reclamation has begun tak-

ing coring samples of the sediment clogged behind the 190-foot-high Matilija Dam in Southern California. By doing so, they can begin to develop and test models as to possible movement, quality, impacts, and aquatic health once the dam comes down. It helps point the way toward the safest, most cost-effective way of getting all that sand from the shallow reservoir back down to the beaches, which have been without it for the past four decades. One emerging possibility is to do this gradually, stage by stage, layer by layer, to minimize impacts to endangered steelhead while opening up their spawning habitat.

This last example, considered the largest dam removal project under way in the world right now, raises a common-sense point that should be made nonetheless: Size matters. The larger the dam, the more extensive the impacts, and thus the more thorough and extensive the scope of preremoval analysis should be. Conversely, there is less reason to do a 5-year environmental impact analysis for removal of a 6-foot-high abandoned dam.

Each example teaches us more about the potential, the possibilities, and, well, the limits of our understanding about dams and dam removal. What works or fails in one place may not apply on another river. By exposing the gaps, we can fill them. By recognizing where there is a need for caution, we can proceed with more confidence. Through documentation and analyses of case studies, we can be guided by the light of science rather than curse the darkness in which we must make projections.

Most recent complex dam removals have proceeded after analysis of potential impacts and consideration of dam proponents' concerns. For the most part, the pressing issues

*Through documentation and
analyses of case studies, we can be
guided by the light of science rather
than curse the darkness in which we
must make projections.*

have been raised in advance, and the right questions have been asked. But have they been well answered? Not as completely or thoroughly as they could be, which highlights why and where we can demand more scientific information. Not just to project impacts and outcomes in advance, for this is being done, but also to confirm that predicted benefits were in fact met and that no negative impacts occurred.

It has been disturbing, looking back, to realize that despite the scale and cost and hype over the past century, almost no postproject scientific analysis was ever done on dam construction. We cannot let that shortcoming extend to dam removal, despite several obstacles.

One obstacle is sheer velocity. What once appeared impossible suddenly seems inevitable. Five years ago, people asked of dam removal, Why? or whether. Society now asks: Which ones, when, and how? Each year that I was with the interior department, I was so busy rushing to champion dam removal events—in Oregon, Washington, Wisconsin, Maine, North Carolina, Pennsylvania, and California—that it was hard to distill the patterns and to follow up to ensure expectations were met. But moving from one project to the next does not mean we cannot revisit those removals to assess and determine whether expectations were met.

Another obstacle is overcoming our instincts. Removal feels so right and makes so much sense to so many: Surely, consensus-based dam removal would heal the hidden wounds that dams inflicted, restore river functions, bring back the anadromous fisheries from coast to coast. There are signs and suggestions that it is doing just that. It is gratifying to learn that, for the first time in many decades, thousands of Chinook swam up Butte Creek past the site of the former McPherrin Dam, Atlantic salmon and striped bass migrated up the Kennebec past the old site of Edwards Dam, vast schools of shad spawned (and were caught by fishermen) up the Neuse River on the outskirts of Raleigh. But even though we have anecdotal evidence of improvements, there is little hard evidence to confirm it. The lack of studies cries out for new research and peer-reviewed papers by experts in social, economic, and ecological fields.

A third obstacle is economic limits—that is, cost. Not one removal I took part in came top-down from Washington, DC. Each opportunity was driven upward, by local necessity—safety, cost, health, imminent extinction, budgets, and litigation. Local forces were the mothers of invention; we adapted our approach, funding, constituency, answers, funding, tools, and management to the unique needs of the watershed in which the dam belonged. That is politically sound but economically difficult. It often proved hard enough to scrape together funds to ensure safe, low-impact removal, let alone to set aside money for postremoval studies.

These obstacles explain our current situation but do not explain it away. However powerful, no force is an adequate,

long-term substitute for clear, science-driven, consensus-based, and transparent written and accountable policy. Decades ago, dams were built to meet certain laudable goals, goals few can object to even in hindsight. But goals are not enough, unless they are met and, more important, shown to have been met. Dam removal, with equally laudable goals and carried out carefully with the best of intentions, cannot neglect the process of collecting and evaluating the evidence to determine whether the goals were met. This process of evaluation is the cornerstone of adaptive ecosystem management.

The lack of studies cries out for new research

and peer-reviewed papers by experts in social,

economic, and ecological fields.

The proper role of science is to light candles in dark corners. It should reveal paths that can guide and improve decisions by society. This is the case in conservation issues like endangered species, forestry, fire, wetlands, and air and water regulations. Yet when it comes to dams, and now to dam removal, all too often, rather than illuminate and lead policy well ahead of us, the academic field follows from behind.

But in issues like the one in your hands, we show that we are learning lessons from our history. Specifically, we are learning from our legal, societal, ecological, hydrological, economical, biological, and conceptual history of both dams and dam removal.

On that note, let us tip our hats here to those groups and foundations and scientists and land managers who not only herald the healing success stories involved with dam removal that we are proud of, but who are also brave enough to highlight the disappointing outcomes that we learn from.

For wherever we act, there is the potential for wounds to be inflicted and mistakes to be made—mistakes of planning, of expectations, of understanding, and of execution. Though its impacts appear far more beneficial than costly, let us still be humble. Dam removal, like dam construction, is not an end unto itself, only a means to an end. It is a means by which humans can live more responsible lives in harmony with creation, a means that requires the illumination of science, ensuring that we look clearly back, and down, before we can truly move forward on solid ground together.

Legal Perspectives on Dam Removal

MARGARET B. BOWMAN

For more than 100 years, America has led the world in dam building—blocking and harnessing rivers for hydropower, irrigation, flood control, water storage, and other purposes. Now, some 75,000 large dams span our nation's waterways and thousands of smaller dams plug our rivers and streams (NRC 1992, AR/FE/TU 1999, USACE 2001a). Although many dams provide important benefits, some no longer serve any significant purpose, or they have negative impacts that are greater than their benefits. In these cases, dam removal is becoming an increasingly attractive option for achieving conservation goals such as river and fisheries restoration, public safety goals such as elimination of unsafe dams, and other community-revitalization goals through increased recreation and green space.

In the past few decades, the United States has also been a world leader in protecting rivers and wildlife from threats such as point source pollution and unsound riverside development. To accomplish this, the United States has developed a series of laws—the Clean Water Act (CWA) and the Endangered Species Act (ESA), for example—designed to stop further damage to our rivers and to the fish and wildlife that depend on them. Today, our increasing interest in dam removal and our strong environmental protection laws are increasingly interacting, with some unexpected results.

Many legal issues are associated with removal of a dam. Decisions about whether or not to remove a dam are often made in the context of regulatory proceedings. In addition, once a decision has been made to remove a dam, federal, state, and local permits are required for the physical removal of the dam from the river. But because many of the laws that are triggered by a dam removal decision focus on environmental *protection*, they are not easily adapted to the environmental *restoration* activities associated with dam removal, and some laws actually discourage environmental restoration efforts.

This article outlines the legal issues associated with both decisions about whether or not to remove a dam and decisions about how to remove a dam. It then examines how imple-

THIS ARTICLE OUTLINES THE LEGAL ISSUES ASSOCIATED WITH DAM REMOVAL AND EXAMINES HOW ENVIRONMENTAL RESTORATION ACTIVITIES SUCH AS DAM REMOVAL FIT INTO THE EXISTING US LEGAL SYSTEM

mentation of environmental restoration activities such as dam removal fits into our existing legal system and how environmental laws may need to evolve to address the increasing interest in environmental restoration.

Legal issues associated with deciding whether to remove a dam

The decision of whether or not to remove a dam is not a centralized decision that is made by one entity. Depending on who owns the dam, what services the dam provides, and the type and significance of the dam's negative impacts, a decision on dam removal can be made by a federal agency, a state agency, or a private dam owner. Although sometimes dam removal is a voluntary undertaking, many dam removal decisions are the result of legal proceedings—either as a formal outcome of the proceedings or through a negotiated settlement associated with the proceedings.

Dam safety proceedings. The most common legal proceedings resulting in dam removal are safety-related inspections of dams at the state level. Most states have dam safety

Margaret B. Bowman (e-mail: mbowman@amrivers.org) is deputy vice president for conservation at American Rivers, 1025 Vermont Avenue, NW, Washington, DC 20005, where she oversees a nationwide program to remove dams that do not make sense. © 2002 American Institute of Biological Sciences.

laws that require periodic state inspections of every dam over a certain size. For example, New Hampshire has jurisdiction over any structure that is more than 1.2 meters (m) tall or has a storage capacity of 2467 m³ or more (NHDES 2001). If a dam has safety problems, the state official usually can issue a notice to the dam owner requiring the owner to address those problems (e.g., State of Massachusetts 2002). Usually the state cannot order the dam to be removed, but it can instead order that the safety problem be eliminated. This provides the dam owner with a choice of either repairing the dam or removing it. Removal of smaller dams often costs less than repairs. In Wisconsin, for example, an examination of small dam removals showed that removal typically costs three to five times less than estimated safety repair costs (Born et al. 1998).

Hydropower dam regulation. Another regulatory arena that has resulted in dam removals is the regulation of hydropower dams by the Federal Energy Regulatory Commission (FERC) pursuant to the Federal Power Act (US Code, title 16, sec. 791 et seq.) (all US Code citations are available online at <http://uscode.house.gov>). Eleven FERC-regulated dams have been removed since 1963 (Emery 2001), with more than 25 currently under consideration.

There are three regulatory avenues for FERC involvement in a dam removal: (1) dam relicensing, (2) dam safety inspections, and (3) the surrender of a dam's operating license.

Hydropower dam relicensing. The first regulatory avenue is through hydropower dam relicensing. All hydropower dams not owned by the federal government must obtain an operating license from FERC, unless the dam has been issued an exemption or is on a nonnavigable river (US Code, title 16, sec. 797[e]). When these 30- to 50-year licenses expire, the dam owner must reapply to FERC to obtain a new license (US Code, title 16, sec. 808). As part of this licensing process, FERC must determine whether issuing a new license is in the public interest, providing equal consideration to power development and nonpower uses of the river (e.g., fish and wildlife habitat, recreation, aesthetics) (US Code, title 16, sec. 797[e]). In 1994, FERC issued a policy statement concluding that it had the authority as part of a relicensing proceeding to deny a relicensing application and to order a dam to be removed if it determines such an action is in the public interest (Project Decommissioning at Relicensing: Policy Statement, 60 Federal Register 339, Code of Federal Regulations [CFR], title 18, sec. 2.24; all CFR citations are available online at www.access.gpo.gov/nara). FERC expressly exercised this dam removal authority once, in their 1997 order requiring removal of the Edwards Dam on the Kennebec River in Maine (Edwards Mfg. Co., 81 FERC 61,225 [1997]). In addition, FERC has used this authority to study the option of dam removal in several cases, such as on the Clyde River in Vermont, where FERC recommended in a 1996 final environmental impact statement that a breached dam be removed as part of a five-dam relicensing (FERC 1996a) (the dam was subsequently

removed pursuant to a settlement agreement), and on the Presumpscot River in Maine, where FERC is currently considering the option of removing three dams as part of a five-dam relicensing (FERC 2001).

FERC relicensing proceedings have also led to dam removal through settlement agreements. Two dams have been removed through relicensing agreements (Emery 2001), with several additional settlements involving dam removal currently undergoing review at FERC. Some of these settlements have included removal of the dam that was the focus of the relicensing. For example, on the White Salmon River in Washington, FERC considered the alternative of removing the Condit Dam and instead ordered installation of fish passage devices (FERC 1996b). However, the dam owner determined that fish passage devices would be more expensive than dam removal, and thus entered into a settlement with intervening parties to remove the dam (PacifiCorp 1999). In addition, several relicensing settlements have included removal of smaller dams in a multidam hydroelectric project or nonhydro dams on tributary streams as mitigation for the ongoing operations of the primary hydropower dams. For example, on the Menominee River in Wisconsin, Wisconsin Electric entered into a comprehensive settlement for the relicensing of eight projects on the Menominee, Michigamme, and Paint Rivers. The parties agreed to support the relicensing (with certain operating conditions) in exchange for Wisconsin Electric removing three tributary dams (Order Issuing Non-Power License to Wisconsin Electric and Approving Decommissioning Plan, 96 FERC 61,009 [2001]).

FERC dam safety authority. The second regulatory avenue for FERC involvement in a dam removal is through dam safety inspections. FERC has the authority to inspect and ensure maintenance of dam safety at all dams under their jurisdiction (CFR, title 18, part 12). These inspections generally occur every 5 years (CFR, title 18, sec. 12.38). As in state dam safety situations, if FERC identifies safety problems at a dam, it will order the dam owner to alleviate the problem. The dam owner may choose to remove the dam rather than make repairs. For example, a FERC safety inspection of Mussers Dam on Middle Creek in Pennsylvania identified significant safety problems, and the dam owner decided it was cheaper to remove the dam than repair it (Order Accepting Surrender of License, Mussers Dam, 64 FERC 62,097 [1993]). At least four FERC-regulated dams have been removed where the cost of safety repairs was a factor in the removal decision (Emery 2001).

Issuance of license surrender order or nonpower license. The third regulatory avenue for FERC involvement in a dam removal is through the surrender of a dam's operating license. Whenever a FERC-licensed dam is slated for removal, FERC must approve the removal through a license surrender order or the issuance of a nonpower license (US Code, title 16, secs. 799, 808[f]). The question of when it is appropriate to use the license surrender versus the nonpower license approach is still evolving at FERC (e.g., APS 2001, PacifiCorp 2001, FERC 2002).

As part of issuing a license surrender or nonpower license, FERC can impose conditions on how the dam is removed. The requirement to obtain a FERC surrender order or nonpower license applies to removals related to dam relicensing and dam safety, as well as to voluntary removals unrelated to safety or relicensing. For example, the licensee of the Grist Mill Dam on the Souadabscook River in Maine received approval from FERC to surrender its license and complete a voluntary dam removal to restore habitat for migratory fish (Order on Surrender of Exemption, Grist Mill Dam, 84 FERC 61,196 [1998]). And FERC issued a nonpower license to Wisconsin Electric for the removal of the Sturgeon Dam in the Upper Menominee River Basin (Order Issuing Non-Power License to Wisconsin Electric and Approving Decommissioning Plan, 96 FERC 61,009 [2001]).

In addition, whenever a dam owner plans to cease generation of hydropower, the owner must obtain a license surrender or nonpower license from FERC. As part of this proceeding, FERC has the authority to order that the dam be removed, even if this is not the intention of the dam owner. In practice, however, when the dam owner does not wish to remove the dam, FERC has to date issued the license surrender or nonpower license without any associated obligation to remove the structure or demonstrate a plan for periodic dam safety maintenance (e.g., Order Accepting Surrender of Exemption, Walker Mill Hydroelectric Project, 91 FERC 62,208 [2000]).

The Endangered Species Act. The third main legal mandate that has resulted in dam removals is the Endangered Species Act (US Code, title 16, secs. 1531–1543). The ESA has never been used to compel dam removal, although it has been used to consider dam removal in a few cases and has in many cases been the impetus for voluntary removals.

Three sections of the ESA have bearing on dam removal decisions: (1) the prevention of jeopardy provisions in section 7, (2) the prohibition of taking a listed species in section 9, and (3) the recovery planning and implementation provisions in section 4(f).

Section 7 jeopardy consultations. Section 7 prohibits federal actions that jeopardize the continued existence of listed species or that destroy or adversely modify critical habitat (US Code, title 16, sec. 1536[a][2]). Critical habitat can include not only habitat currently occupied by the species but also habitat not currently occupied but “essential for the conservation of the species” (US Code, title 16, sec. 1532[5][A][ii]).

If an activity might result in jeopardy, the federal actor must consult with the US Fish and Wildlife Service (USFWS) or the National Marine Fisheries Service (NMFS). *Jeopardy* means threatening either survival or recovery of the species (see *Sierra Club v. Fish and Wildlife Service*, 245 F.3d 434 [5th Cir. 2001]). As a result of the consultation, NMFS or USFWS will issue a biological opinion determining whether jeopardy will result from the proposed action and recommending “reasonable and prudent alternatives” that can be taken to avoid jeopardy (US Code, title 16, sec. 1536[b][3]). ESA reg-

ulations mandate that reasonable and prudent alternatives be implementable in a manner consistent with the original project purposes and be within the legal authority of the federal actor (CFR, title 50, sec. 402.02). If no reasonable and prudent alternative exists, NMFS or USFWS must issue a jeopardy opinion with no reasonable and prudent alternative. At this point, an application for exemption from the provisions of the ESA could be made to the Endangered Species Committee (or “God Squad”) (US Code, title 16, sec. 1536[g]). In determining whether exemption is warranted, the God Squad may consider “alternative courses of action” that are not limited to original project purposes (US Code, title 16, secs. 1532[1], 1536[h]). The God Squad provision has been treated as a legal and political last resort, being used in only a very small number of cases (Weston 1993).

If a dam is threatening the continued survival or recovery of a species, and if the dam is not central to the purpose of the project and removal is within the authority of the federal actor, the ESA may authorize USFWS or NMFS to issue a jeopardy opinion that recommends removal of the dam. NMFS has recommended in a section 7 biological opinion the notching of a half-constructed dam (the Elk Creek Dam in Oregon) as the only alternative that would avoid jeopardy (NMFS 2001) and has in at least one other biological opinion (regarding the Eel River’s Potter Valley Project in California) recommended studying dam removal for salmon protection (NMFS 2000a). However, the Eel River dam removal study recommendation was not made as part of the biological opinion’s reasonable and prudent alternatives, but instead as part of the less enforceable recommended conservation measures. In addition, NMFS has considered—and temporarily rejected—dam breaching as an option for salmon protection and restoration in its 2000 biological opinion regarding four federal dams on the Lower Snake River in Washington (NMFS 2000b).

The use of section 7 to mandate removal has been problematic in several ways, however:

- First, section 7 applies only to actions taken (or licensed) by the federal government. Thus if there is no federal actor, this section will not apply.
- Second, section 7 is triggered only by a proposed action, and it can be a challenge to characterize the continued existence of a dam as a proposed action. In the case of the Snake River dams, the federal government’s annual operating plan for the dams has been sufficient to trigger section 7 consultation (e.g., *Idaho Department of Fish and Game, et al. v. National Marine Fisheries Service*, 56 F.3d 1071 [9th Cir. 1995]). However, in other situations, it is not settled whether section 7 consultation must be initiated for ongoing federal activities. For example, FERC has ruled that section 7 consultation obligations are not triggered by provisions in FERC licenses that allow FERC to reopen the license if necessary to protect fish and wildlife (Order Dismissing Conservation Groups’ Request for Rehearing re Puget Sound Energy, Inc., under P-2150, 95 FERC 61,319

[2001]), but this conclusion is currently on appeal in federal court (*Washington Trout, Washington Environmental Council and American Rivers v. Federal Energy Regulatory Commission*, case no. 01-71307 [US Court of Appeals, 9th Cir., filed 30 July 2001]).

- Third, another obstacle became apparent with the 2000 biological opinion for the Snake River dams: It can be hard to demonstrate not only that a dam jeopardizes the continued existence of an entire species, but also that dam removal is necessary to avoid jeopardy.
- Fourth, reasonable and prudent alternatives must be consistent with the original project purposes. Because dam removal usually eliminates the uses of the dam, it may be difficult for NMFS or USFWS to recommend dam removal unless the dam is not central to the project's purposes.
- Fifth, although the ESA enables designation of critical habitat that is currently unoccupied (such as fish habitat above a dam where the dam has no fish passage), section 7 may only prevent destruction or adverse modification of the habitat; it is currently unsettled whether it could also require or promote restoration of critical habitat. Thus where important spawning or rearing habitat for a listed species is flooded by a dam's reservoir, it is unclear whether section 7 could be used to mandate dam removal to restore that habitat.

Section 9's prohibition on taking listed species. Section 9 of the ESA forbids all persons from taking a listed species (US Code, title 16, sec. 1538). The act defines *take* as "to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct" (US Code, title 16, sec. 1532[19]). Harm to the listed species' habitat may also constitute a take (*Babbitt v. Sweet Home Chapter of Communities for a Great Oregon*, 515 US 687 [1995]). To clarify the differences between section 9 and section 7 obligations, section 9 was designed to prevent an individual from fishing for and killing an endangered fish, though it also can apply to broader situations, such as the killing of fish in a dam's turbines. In contrast, section 7 was designed to address threats to the whole species, such as eliminating all fish passage on a river through construction of a dam.

As an exception to the section 9 prohibition on taking a species, the ESA allows USFWS or NMFS to permit "incidental" take when the proposed activity is not likely to jeopardize continued existence of the species and when the taking of species is not the purpose of the action. These incidental take permits may be issued only for federal actors in conjunction with a biological opinion issued pursuant to section 7 (called "incidental take statements") and for nonfederal actors in conjunction with a habitat conservation plan developed pursuant to section 10(a) (US Code, title 16, secs. 1536[b][4], 1536[o][2], 1539[a][2][A]).

Section 9 applies to all actors, and it applies when (for example) only one fish is affected, not just (as with section 7) when the action might jeopardize the existence of the whole

species. Section 9 could authorize USFWS or NMFS to declare continued operations of a dam an impermissible taking where the dam's operations injure or kill listed fish. To enforce this finding, USFWS and NMFS could only issue fines, but a citizen suit to enforce a takings finding could result in an injunction (US Code, title 16, sec. 1540; *Marbled Murrelet et al. v. Pacific Lumber Co.*, 83 F.3d 1060 [9th Cir. 1996]). Where listed fish are currently using fish passage devices at a dam with a resulting mortality rate, the case that the dam is causing a take is relatively straightforward. However, if a dam is currently a complete block to fish passage (with no fish kills below the dam), making a case that the dam causes a take may be more challenging. NMFS has initiated take proceedings seeking dam removal only once—at the Savage Rapids Dam on Oregon's Rogue River, where ineffective fish passage is harming threatened coho salmon (*National Marine Fisheries Service v. Grants Pass Irrigation District*, no. 98-3034-HO [D.Or. filed 22 April 1998]). As part of a comprehensive settlement of both the Savage Rapids take proceedings and an associated state court water rights adjudication, NMFS issued a 1-year incidental take permit justified by the planned removal of the dam (NOAA 2001).

ESA's recovery planning and implementation obligations. The ESA also requires USFWS and NMFS to develop and implement recovery plans for "the conservation and survival" of threatened and endangered species unless the agency finds that "such a plan will not promote the conservation of the species" (US Code, title 16, sec. 1533[f]). It also requires all federal agencies to carry out programs aimed at recovery and requires USFWS and NMFS to use all programs they administer to further conserve the species (US Code, title 16, sec. 1536[a][1]). These provisions can be interpreted to provide authority to NMFS and USFWS to develop and implement species recovery plans that include dam removal and to require other agencies to follow those plans. However, this has not happened to date. In practice, the recovery planning and program administration obligations in the act have generally not been enforceable (Cheever 1996), and because of funding and political constraints, recovery plans are not always developed. (Of the 1244 listed species in the United States as of 31 July 2001, recovery plans have been developed for only 975 species [USFWS 2001]).

ESA as a factor in other dam removal decisions. Despite the fact that no dams have been ordered to be removed under ESA authority, the presence of listed species at a dam (particularly fish) has been a significant factor in many decisions to remove dams. This includes voluntary dam removals, such as on Clear Creek in California, where the Saeltzer Dam was removed in 2000 to restore habitat for threatened and endangered salmon and trout species (Hepler 2001), as well as formal proceedings to determine whether a dam should be removed, such as the CalFed Bay-Delta Program's consideration of removing Englebright Dam on the Yuba River in California to restore chinook salmon and steelhead (FOTR 1999). In fact, all seven dam removals in the Pacific Northwest and

California in 1999, 2000, and 2001 were conducted to restore endangered fishes (American Rivers 2002).

Obtaining permits to remove a dam

Removing a dam from a river requires permits from state, federal, and local authorities. These permits are generally required to ensure that the removal is done safely and minimizes short- and long-term impacts to the river and riparian area. Although most states have the same basic categories of permits required for a dam removal, there is substantial variation from state to state in the level of review required and the standards that must be met to permit a dam removal. In some states, dam removal permitting is relatively easy, and in other states, it is difficult. Below is a summary of the types of federal, state, and local permits that may be required for removal.

Federal permits or requirements.

Clean Water Act section 404 permit. Most dam removals require a CWA section 404 permit, issued by the US Army Corps of Engineers (Corps) for dredging of a navigable waterway (US Code, title 33, sec. 1344). A guideline pursuant to this statutory requirement establishes a policy of no net loss to wetlands (EPA and Department of the Army 1990). To obtain Corps approval, the project (a) should not cause or contribute to significant degradation of the waters or result in a net loss of wetlands, (b) should be designed to have minimal adverse impact, (c) should not have any practicable alternatives, and (d) should be in the public interest. In some cases, dam removal will result in a net loss of wetlands. To obtain a permit in these situations, the Corps will have to find that the benefits of dam removal outweigh the loss of wetlands, or that the loss of wetlands are mitigated by creation of wetlands elsewhere. In October 2001, the Corps issued a regulatory guidance letter that permits mitigation of wetlands impacts with nonwetland habitats (USACE 2001b). Other federal agencies are currently commenting on this letter, and it remains to be seen whether the letter effectively abandons the policy of no net loss of wetlands.

Rivers and Harbors Act permit. In conjunction with a CWA section 404 permit, the Corps will issue a Rivers and Harbors Act section 10 permit (US Code, title 33, sec. 403). The Rivers and Harbors Act is administered by the Corps for federal activities affecting a navigable waterway. The Corps will issue the permit if there is no adverse impact on interstate navigation.

FERC license surrender or nonpower license approval. If the dam to be removed is a FERC-regulated hydropower dam, the dam owner will have to apply for surrender of the FERC license or issuance of a nonpower license, as discussed in the section "Hydropower dam regulation," above.

National Environmental Policy Act (NEPA) review. A permitting or licensing action by the Corps or FERC may require the preparation of an environmental impact statement or environmental assessment pursuant to NEPA (US Code, title 42, sec. 4321 et seq.). A NEPA environmental document

may already have been prepared as part of the process of deciding whether to remove the dam. If this is the case, it may not be necessary to prepare a new NEPA document, or only a supplemental document may be required.

Federal consultations. As part of issuing their permits, the Corps or FERC may need to conduct the following consultations:

- **ESA section 7 consultation.** If threatened or endangered species are present at or near the dam, the Corps or FERC may need to consult with USFWS or NMFS regarding the impact of the removal on these species, as discussed above in the section "The Endangered Species Act."
- **Magnuson-Stevens Act consultation.** The Corps and FERC may also need to consult with NMFS pursuant to the Magnuson-Stevens Act regarding the impact of the removal on any fishery management plan developed by a regional fishery management council (US Code, title 16, sec. 1855[b][2]). This consultation is done to ensure that the removal will not adversely affect any essential fish habitat established in the fishery management plan.
- **National Historic Preservation Act consultation.** Corps or FERC activities may also trigger an obligation to assess the impact of the proposed action on historic properties pursuant to section 106 of the National Historic Preservation Act (US Code, title 16, sec. 470[f]). In assessing this impact, FERC or the Corps must consult with the state historic preservation officer. Affected historic properties may range from newly exposed archaeological sites to the dam itself. The presence of a dam on the National Register of Historic Places (or eligibility for listing on the register) does not automatically preclude removal. In many situations, proper documentation of the dam before removal may be sufficient to preserve the historic values of the dam (CFR, title 36, sec. 800.1 et seq.).

State certifications. The Corps and FERC decisions also trigger several federal statutes that require the state to issue a certification that the actions are consistent with the state's implementation of federal law.

- **Water-quality certification.** For the Corps to issue a CWA section 404 permit or for FERC to issue a license surrender order or nonpower license, the state must issue a water-quality certification pursuant to CWA section 401 (US Code, title 33, sec. 1341). This certification states that the proposed activity will not result in the violation of state water-quality standards. The state may issue conditions for how the dam should be removed as part of its certification.
- **Coastal Zone Management Act certification.** If the dam is located in a coastal zone, in order for the Corps or FERC to permit the dam removal, the state must issue a certification pursuant to the Coastal Zone Management Act (US Code, title 16, sec. 1451 et seq.). This certification states that the proposed activity is consistent with the state's approved coastal zone management program.

Again, the state may issue conditions for how the dam should be removed as part of its certification.

State permits.

Waterways development permits. Some states have laws that regulate the development of their waterways for hydropower, navigation, and other purposes. These laws are generally adopted to address construction of a new dam or alteration of an existing dam but may also apply to dam removal.

Dam safety permits. Most states have regulations that require a permit for any activity that will affect the safety of a dam. Removal of a dam may require such a permit.

State environmental policy act review. Many states have an environmental impact review statute similar to the federal NEPA statute. The removal of a dam may trigger the state requirement to prepare an environmental impact document. Usually the federal and state requirements can be met by preparing the same environmental impact document.

Historic preservation review. Most states require that before any state permit is issued, historic and archaeological issues must be investigated and approved by the state historic preservation officer. This review can usually be done in conjunction with the federal historic preservation review, described earlier.

Resetting the floodplain. Most states will require a review of any activity that might change the 100-year floodplain. The applicant may be required to determine the new elevation for the 100-year floodplain once the dam is gone. The Federal Emergency Management Agency would then use the analysis to create new maps.

State certifications. State certification requirements pursuant to federal laws are discussed above, under "Federal permits or requirements."

Municipal permits. The act of demolishing the structure of the dam may require a demolition permit from the local municipality, and the construction of a temporary cofferdam or the restoration of the riverbank may require a building permit from the local municipality.

Legal impediments to ecological restoration

Environmental laws protect against deviations from the status quo. Environmental laws in the United States focus primarily on environmental protection. Recently, however, there has been an evolution of interest in environmental science and activism from protection to restoration. In many areas, the legal system has not kept up with this evolution. Many environmental laws have protection and restoration goals. For example, the stated goal of the Clean Water Act is to "restore and maintain the chemical, physical and biological integrity of the Nation's waters" (US Code, title 33, sec. 1251), and the goal of the ESA also focuses on recovery of listed species (US Code, title 16, secs. 1531[b], 1532[3]). But

environmental laws effective at environmental protection (such as the CWA and ESA) are essentially effective only at maintaining the status quo. For example, the Clean Water Act's most effective provisions are focused on preventing pollution from entering rivers and other waterways, and implementation of the Endangered Species Act is focused primarily on preventing further degradation of an endangered species (Cheever 1996 discusses how ESA implementation focuses on the act's prohibitions and not on its purpose). Unlike environmental protection efforts, environmental restoration projects such as dam removal result in a deviation from the status quo (albeit positive). As a result, where laws focus on preventing deviations from the status quo to meet their protection goals, they can actually discourage restoration activities.

Dam removal is a good example of this problem. Although dams are being removed to accomplish ecological restoration goals, these removals are often being accomplished in spite of environmental laws designed to protect those resources. Instead, the decision to remove a dam may be accomplished through laws designed to allow a balancing of interests and negative deviations from the status quo, such as hydropower dam relicensing pursuant to the Federal Power Act and state dam safety laws.

Dam removal is not the only situation where this dichotomy exists. For example, the effort to reoperate the Glen Canyon Dam on the Colorado River to restore the health of the river through the Grand Canyon has met several regulatory obstacles designed to stop environmental degradation (Schmidt et al. 1998, Miller 2000). The everglades restoration effort has also encountered challenges from environmental protection laws (Rizzardi 2000).

Example: The Edwards Dam on the Kennebec River. The removal of the Edwards Dam on the Kennebec River in Maine provides a good example of this dichotomy. Built in 1837, Edwards Dam blocked the migration route for seven target species of anadromous fish—Atlantic salmon (*Salmo salar*), striped bass (*Morone saxatilis*), American shad (*Alosa sapidissima*), alewife (*Alosa pseudoharengus*), rainbow smelt (*Osmerus mordax mordax*), Atlantic sturgeon (*Acipenser oxyrinchus*), and endangered shortnose sturgeon (*Acipenser brevirostrum*). The dam also flooded unique head-of-tide habitat important for the life cycles of many of the migratory fish. The dam's license to generate power expired in 1993, and the dam owners sought a new 30-year license from FERC. In response, four environmental groups and state and federal resource agencies intervened in the licensing to seek dam removal.

After a long regulatory battle, in 1997 FERC denied the dam owner's application for license renewal and, for the first time ever, ordered the dam to be removed against the wishes of its owner (Edwards Mfg. Co., 81 FERC 61,225 [1997]). Pursuant to a subsequent settlement agreement, the dam was removed in 1999. Today, the former impoundment has been restored to a healthy river ecosystem that supports a diverse array

of fish and wildlife, including the seven target anadromous fish species (NRCM 2001).

Although there were compelling environmental reasons to remove Edwards Dam, environmental laws provided little if any leverage to remove the dam—they actually created some challenges for designing and permitting the removal. The Edwards removal involved two decision points where environmental laws came into play: the decision whether to order dam removal and the permitting of the removal itself.

The dam removal decision. The most significant environmental law involved in the dam removal decision was the Endangered Species Act. The shortnose sturgeon—a federally listed endangered species—was present below the dam and historically migrated upstream above Edwards to spawn in the impoundment area. The relicensing proceeding required FERC to consult with USFWS and NMFS pursuant to ESA section 7. But the ESA provided no legal tools to promote dam removal. No critical habitat had been designated for the sturgeon, and no recovery plan had ever been developed. The Edwards Dam itself did not jeopardize continued existence of the shortnose sturgeon; it was simply inhibiting the species' recovery. However, section 7 simply creates an obligation not to destroy existing habitat. It has not been used to require restoration of historic habitat. In addition, even if USFWS and NMFS had developed a recovery plan under section 4(f) that called for removal of Edwards Dam to restore historic habitat, it still would have been difficult to mandate removal pursuant to the plan. FERC was the decisionmaker in the Edwards case, and FERC has no recovery obligation under section 4(f) of the ESA. Instead, an argument would have to be mounted that FERC's ESA section 7(a)(1) obligation to carry out programs aimed at recovery mandates that FERC follow the USFWS and NMFS recovery plan and order dam removal. Whether section 7(a)(1) is enforceable in this manner is unsettled, though a majority of courts have rejected these claims (Cheever 1996).

In addition, pursuant to CWA section 401, the state of Maine was charged with certifying whether the licensing would violate state water-quality standards. Removal of the dam would probably improve water quality, and the state's denial of certification would have prevented FERC from issuing a new license. Section 401 certification conditions regarding dam relicensings traditionally require actions that prevent further degradation of numeric water-quality standards (such as increased downstream flows to prevent dissolved oxygen violations), though states have increasingly been imposing non-status quo actions, such as building fishways to meet descriptive water-quality standards or designated uses (such as restoring native fish populations to river stretches designated as habitat for native fish). Although the state supported removal of the dam, it felt that it had no avenue through its Clean Water Act authority to mandate removal to improve numeric water-quality conditions above the dam, though it did recommend fish passage to ensure native fish access to historic spawning grounds (State of Maine 1996). In the end, the Edwards Dam removal resulted in significant

improvement to the Kennebec's water quality—the former impoundment area changed from failing to meet Maine's minimum water-quality standard before dam removal to attainment of Class B standards within 2 months after removal (NRCM 2001).

Finally, as part of the FERC relicensing process, the USFWS and NMFS have authority to recommend conditions on a proposed license pursuant to the Federal Power Act and the Fish and Wildlife Coordination Act (US Code, title 16, sec. 661 et seq.). Although NMFS and USFWS may submit any recommended license conditions for FERC's consideration, the two agencies are granted authority to impose mandatory conditions for construction of fishways—FERC must include the USFWS and NMFS conditions in the license (US Code, title 16, sec. 811). Although the purpose of NMFS's and USFWS's involvement in FERC relicensings includes "wildlife conservation and rehabilitation" (US Code, title 16, sec. 661), they are limited to mandating fishways to enable passage at the dam—they cannot mandate dam removal even if that is the only way to achieve fish passage. In the Edwards Dam case, USFWS and NMFS had concluded that fishways would not be effective at passing the target fish species, and that dam removal was the only way the target fish species could be restored. Nevertheless, the only action they could mandate to provide fish passage at the dam was construction of fishways. Thus the agencies *recommended* dam removal, but *ordered* construction of fishways (e.g., NMFS 1996).

In the end, no environmental law provided sufficient authority to remove Edwards Dam. Instead, a nonenvironmental law—the Federal Power Act—was used to obtain an order to remove the dam. FERC's relicensing decision pursuant to the Federal Power Act was based on the economic conclusion that construction of fish passage devices would cost 1.7 times more than dam removal and on the biological conclusion that even if a fish passage device were constructed, it could be used by only three of the seven target fish species (Edwards Mfg. Co., 81 FERC 61,225 [1997]). (American Rivers [2001] provides further information about the FERC relicensing process that led to dam removal.)

Obtaining permits for removal. In addition to obtaining an order from FERC to remove Edwards Dam, project proponents also were required to obtain permits to carry out the removal, as described in the section "Obtaining permits to remove a dam," above.

Obtaining a CWA section 404 permit for the removal triggered a second obligation pursuant to ESA section 7 to consult with NMFS and USFWS about impacts to the shortnose sturgeon. Immediately below the dam was a large scour hole created by water flowing over the dam. The sturgeon used this hole for spawning because they were no longer able to move upstream to their historic spawning holes above the dam. Upon removal of the dam, it was expected (and it came to pass) that this hole would be filled in by debris and coarse sediments transported downstream. Although this spawning hole would be lost, access to the sturgeon's historic spawning areas would be reopened through dam removal. If the spawn-

ing hole below the dam had been formally designated as critical habitat for the shortnose sturgeon, dam removal may have been hard to accomplish because of the complete destruction of this critical habitat. However, critical habitat had never been designated for the shortnose sturgeon on the Kennebec. Thus although there was concern about the loss of the spawning hole, a formal conflict with ESA on this issue did not exist.

ESA section 9 also created challenges for the Edwards Dam removal. If removal of Edwards Dam harmed or killed any of the shortnose sturgeon residing in the river, it would have been in violation of section 9's prohibition against taking of an endangered species. The timing and method of the removal was substantially changed to avoid violation of this provision.

Conclusions

As the Edwards Dam removal illustrates, existing laws that are effective at ensuring environmental protection will probably not be effective at promoting environmental restoration activities such as dam removal. The resulting question is how to allow positive deviations from the environmental status quo while not weakening laws and creating loopholes that will allow more negative deviations from the status quo. Basic exemption from environmental protection laws for restoration projects is not advisable, because environmental restoration projects do have impacts that need to be reviewed and minimized.

A better approach may be to provide regulatory direction or guidance that allows a decisionmaker to provide some accommodation for projects with restoration as their primary purpose. For example, a state or federal agency could establish a policy that enables flexibility in the interpretation of permitting requirements when a proposed project's primary purpose is environmental restoration. An agency could also direct permitting officials to consider the long-term benefits of a restoration project as mitigating factors in determining whether the short-term impacts of the project are acceptable. The challenge is to develop this in a fashion that avoids the appearance (or reality) of unfair treatment or relies so heavily on professional judgment that it renders the regulations unpredictable or unenforceable. And if restoration activities are given special accommodation, it will be especially important that the project proponents demonstrate that the restoration goals were actually met.

In addition to enabling existing laws to accommodate restoration in a more effective manner, these laws should be able to meet their goals of actively promoting environmental restoration. The experience to date indicates that this has been either legally or politically difficult. It remains to be seen whether the increasing attention to restoration in the scientific and activist communities will help move implementation of environmental laws toward their restoration goals or instead demonstrate the need for new legislation dedicated to environmental restoration.

Acknowledgments

More than a dozen colleagues reviewed drafts and otherwise assisted with this article. Although they are too numerous to mention by name, they have my sincere appreciation. I am particularly grateful to Ron Kreisman, Steve Brooke, Peter Kirsch, and Dan Squire, who for years explored the legal complexities of dam removal with me on the Edwards Dam relicensing. Finally, my thanks to The Pew Charitable Trusts for their support for the development of this article.

References cited

- American Rivers. 2001. A history of Edwards Dam removal efforts: The FERC relicensing process. (16 July 2002; www.amrivers.org/tableofcontents/history.htm)
- . 2002. Dams removed in 1999, 2000 and 2001. (16 July 2002; www.amrivers.org/damremoval/damremovals2001.htm)
- [AR/FE/TU] American Rivers, Friends of the Earth, Trout Unlimited. 1999. Dam Removal Success Stories: Restoring Rivers through Selective Removal of Dams That Don't Make Sense. Washington (DC): AR/FE/TU.
- [APS] Arizona Public Service Co. 2001. Petition for declaratory order and supplemental comments on offer of settlement (P-2069). 1 August. FERC submittal 20010810-1370.
- Born S, Genskow K, Filbert T, Hernandez-Mora N, Keefer M, White K. 1998. Socioeconomic and institutional dimensions of dam removals: The Wisconsin experience. *Environmental Management* 22: 359–370.
- Cheever F. 1996. The road to recovery: A new way of thinking about the Endangered Species Act. *Ecology Law Quarterly* 23: 1–78.
- Emery L. 2001. A review of nonfederal hydropower dams removed under FERC's regulatory authority. Paper presented at Waterpower XII Conference; 9–11 July 2001; Salt Lake City, UT.
- [EPA] Environmental Protection Agency, Department of the Army. 1990. Memorandum of agreement between the EPA and the Department of the Army concerning the determination of mitigation under the Clean Water Act section 404(b)(1) guidelines. 6 February.
- [FERC] Federal Energy Regulatory Commission. 1996a. Final Environmental Impact Statement, Clyde River Hydroelectric Project, Vermont (P-2306). Washington (DC): FERC.
- . 1996b. Final Environmental Impact Statement, Condit Hydroelectric Project, Washington (P-2342). Washington (DC): FERC.
- . 2001. Draft Environmental Impact Statement, Presumpscot River Projects, Maine (P-2942, 2931, 2932, 2941, 2897). Washington (DC): FERC.
- . 2002. Supplemental Environmental Impact Statement, Condit Hydroelectric Project, Washington (P-2342). Washington (DC): FERC.
- [FOTR] Friends of the River. 1999. Rivers Reborn: Removing Dams and Restoring Rivers in California. Sacramento (CA): FOTR. (13 July 2002; www.friendsoftheriver.org/pub/RiversReborn.pdf)
- Hepler TE. 2001. Ten miles to Whiskeytown—The removal of Saeltzer Dam. Pages 545–559 in US Society on Dams, ed. *The Future of Dams and Their Reservoirs*. Denver (CO): US Society on Dams.
- Miller SK. 2000. Undamming Glen Canyon: Lunacy, rationality, or prophecy? *Stanford Environmental Law Journal* 19: 121–205.
- [NHDES] New Hampshire Department of Environmental Services, Dam Bureau. 2001. New Hampshire's Definition of a Dam. Concord (NH): NHDES. (26 July 2001; www.des.state.nh.us/dam/damurweb.pdf)
- [NMFS] National Marine Fisheries Service. 1996. Final Section 18: Fishway Prescription, Edwards Dam, P-2389. Silver Springs (MD): NMFS. FERC submittal 19961010-0147.
- . 2000a. Draft Biological Opinion for the Proposed License Amendment for the Potter Valley Project (FERC no. 77-110). Silver Springs (MD): NMFS.
- . 2000b. Biological Opinion, Reinitiation of Consultation on Operation of the Federal Columbia River Power System, including the Juvenile Fish Transportation Program and 19 Bureau of Reclamation Projects in the Columbia Basin. Silver Springs (MD): NMFS.

- . 2001. Biological Opinion and Magnuson-Stevens Act Essential Fish Habitat Consultation, Elk Creek Dam. Silver Springs (MD): NMFS.
- [NOAA] National Oceanic and Atmospheric Administration. 2001. Issuance of Permit 1307, Grants Pass Irrigation District. Washington (DC): NOAA. Federal Register 66: 38,417.
- [NRC] National Research Council. 1992. Restoration of Aquatic Ecosystems: Science, Technology, and Public Policy. Washington (DC): National Academy Press.
- [NRCM] Natural Resources Council of Maine. 2001. Environmental Benefits of Edwards Dam Removal. (17 July 2002; www.maineenvironment.org/Edwards_Dam/AnniversaryFacts.htm)
- PacifiCorp. 1999. Application for amendment of license and approval of offer of settlement re Condit Project 2342. FERC submittal 19991025-0124.
- . 2001. Petition for declaratory order re Condit Project 2342. FERC submittal 20010605-0193. 1 June.
- Rizzardi KW. 2000. The Everglades in jeopardy: A drama of water management and endangered species. *Florida State University Law Review* 27: 349–395.
- Schmidt JC, Webb RH, Valdez RA, Marzolf GR, Stevens LE. 1998. Science and values in river restoration in the Grand Canyon. *BioScience* 48: 735–747.
- State of Maine. 1996. Comments of State of Maine on Kennebec River Basin DEIS and initial comments, recommendations, terms and conditions on revised application for Edwards Dam, P-2389. FERC submittal 19960409-0094. 5 April.
- State of Massachusetts. 2002. General Laws of Massachusetts, chapter 253, sections 46, 47. (14 July 2002; www.state.ma.us/legis/laws/mgl/gl-253-toc.htm)
- [USACE] US Army Corps of Engineers. 2001a. National inventory of dams. (26 July 2001; www.tec.army.mil/nid/index.html)
- . 2001b. Guidance for the establishment and maintenance of compensatory mitigation projects under the Corps Regulatory Program Pursuant to section 404(a) of the Clean Water Act and Section 10 of the Rivers and Harbors Act of 1899. Regulatory Guidance Letter 01-1. 31 October.
- [USFWS] US Fish and Wildlife Service. 2001. Threatened and Endangered Species System. Washington (DC): US Fish and Wildlife Service. (31 July 2001; www.ecos.fws.gov/tess/html/boxscore/html)
- Weston JL. 1993. Comment: The endangered species committee and the northern spotted owl: Did the "God Squad" play God? *Administrative Law Journal (American University)* 7: 779–817.

A Geomorphic Perspective on Nutrient Retention Following Dam Removal

EMILY H. STANLEY AND MARTIN W. DOYLE

Streams and rivers of Wisconsin reflect the influence of more than 100 years of human activity. These systems have been subjected to upland and channel alterations that include deforestation, wetland drainage, soil inputs from poor farming practices, dam construction, and nutrient enrichment from point and nonpoint sources. Agricultural activities in particular have influenced water quality through modifications such as fertilizer application, increased upland erosion, ditching and tile draining to move water off the land, and straightening of channel ways. The combined effect of nutrient loading and simplification of the physical structure of agricultural streams is to diminish the ability of these systems to retain nutrients (Royer et al. 2001). Because the availability of nitrogen (N) or phosphorus (P) (or both) often limits rates of biological processes in aquatic systems, recent increases in delivery of N and P to lakes, streams, and rivers have acted to fertilize not only the receiving freshwater ecosystem but also coastal areas, resulting in undesirable increases in productivity in both freshwater and marine systems (Carpenter et al. 1998, NRC 2000).

The second conspicuous human influence on streams and rivers of the state is the widespread presence of dams. There are approximately 3700 dams, or 1 dam every 14 kilometers of river in Wisconsin (WDNR 1995). Although there is a healthy representation of large dams (structures > 2 meters [m] that impound $\geq 62,000$ m³, or structures > 7.6 m that impound 18,500 m³; USACE 1998), state waterways are more commonly populated by high densities of small, run-of-river structures, many of which are well over 80 years old. (Run-of-river structures are dams that create reservoirs with small storage capacity and do not alter the river's flow regime.) The abundance of dams can be traced back to the Milldam Act of 1840, which encouraged the use of hydropower to fuel the state's burgeoning economy (Martini 1998). Unfortunately, many of these structures are no longer economically viable, represent a safety risk, and compromise the quality of

GEOMORPHIC MODELS PROVIDE A MEANS OF PREDICTING ECOSYSTEM RESPONSES TO DAM REMOVAL

the aquatic resource (Born et al. 1998). Under these circumstances, dam removal is a logical management option, and more than 50 dams have been removed under the supervision of the Wisconsin Department of Natural Resources over the past three decades. To date, and reflective of the national trend (Doyle et al. 2000), most dams that have been removed within the state were relatively small structures.

The growing interest and occurrence of dam removal underscores the inextricable link between agriculture and river modification in the Midwest. Dams were often built for milling of agricultural products, and the sediment-trapping ability of reservoirs means that topsoil and nutrients lost from farm fields are now stored behind dams. Given the growing concerns about nutrient enrichment and the potential for dam removal to affect nutrient dynamics, understanding the effects of dam removal on nutrient processes should be a research and management priority. In this article, we draw from the context of agriculturally dominated watersheds in Wisconsin to explore how dam removal may influence the movement of N and P in rivers. We approach this issue first by briefly considering nutrient transport in rivers and how reservoirs can affect nutrient processes. We then consider

Emily H. Stanley (e-mail: ehstanley@facstaff.wisc.edu) is an assistant professor of river ecology in the Center for Limnology at the University of Wisconsin, Madison, WI 53706. Martin W. Doyle (e-mail: mwdoyle@email.unc.edu) is an assistant professor in the Department of Geography at the University of North Carolina, Chapel Hill, NC 27599. © 2002 American Institute of Biological Sciences.

changes in nutrient dynamics following dam removal with respect to geomorphic adjustments caused by the removal. Although the specific geomorphic changes that can occur following dam removal will vary among stream types (Pizzuto 2002), the example presented herein emphasizes the link between geomorphic adjustments and ecosystem responses following dam removal.

Nutrient retention in streams and rivers

The increase in nutrient concentrations in many aquatic systems over the past several decades is now well established (Carpenter et al. 1998, NRC 2000). Typically, N and P inputs to aquatic ecosystems are dominated by diffuse nonpoint sources from the surrounding landscape, often in association with agricultural and urban land uses. In many areas, including Wisconsin, fertilizer and manure application on farm fields represents a major input of both N and P to lakes and streams. But the paths that these two nutrients take from terrestrial to aquatic environments are distinct. In enriched systems, nitrate (NO_3^-) represents the dominant form of N, often accounting for more than 50% of the total N budget (Hedin et al. 1995, Goolsby et al. 1999). This form of N is highly soluble and thus travels easily in water from soil to groundwater and into surface water systems. It is also readily taken up by algae and bacteria, which can lead to excess growth of these microorganisms in aquatic systems. Fortunately, NO_3^- can be removed from water and returned to the atmosphere via the process of denitrification—that is, the conversion of NO_3^- to a gaseous and relatively inert form of N (N_2) by bacteria. This transformation occurs under conditions in which the oxygen (O_2) is absent or its concentration is reduced, such as when NO_3^- -rich groundwater travels through wetland soils or streambed sediments. For streambeds, sediment composition plays a key role in determining whether or not denitrification can occur; streambeds with a coarse gravel substrate have a lower potential for denitrification than those with finer sediments, because oxygen concentrations generally remain high in porous sediments (Garcia-Ruiz et al. 1998). Thus an important aspect of NO_3^- (and thus total N) retention in streams and rivers is the degree to which NO_3^- -rich water encounters areas in which O_2 is depleted (NRC 1992).

In contrast to the high mobility of NO_3^- , P movement through ecosystems is relatively slow and is dependent on erosion and sediment transport. The inorganic form of P (phosphate, PO_4^{3-}) has a high affinity for mineral surfaces and therefore easily attaches to sediment and soil particles. Phosphorus fertilizer applied to farm fields typically stays in place and slowly builds up over time; its transfer to the aquatic environment requires mobilization and transport of soil particles. The combination of farming and urban development has fostered both a widespread buildup of P in soils and the transport of these soils to aquatic systems (Bennett et al. 2001). Reservoirs and lakes then trap particles and store this legacy of fertilization and land use for years. Thus while movement of N is strongly influenced by the extent of inter-

actions between NO_3^- -rich water and O_2 -poor sediments (or sediment–water contact), P transport is often driven by the movement of particles in streams and rivers, particularly in sediment-rich systems characteristic of basins with substantial agricultural land use (Ng et al. 1993).

It has long been assumed that once N and P enter a stream, fluvial systems do little more than transport the nutrients to downstream environments. While the idea of streams as transporters is still pervasive, the nutrient spiraling concept (Webster and Patten 1979, Newbold et al. 1981) has emphasized the role of streams as transformers as well as transporters of elements such as N and P. With the awareness that streams and rivers can remove and transform nutrients and materials as well as transport them, questions now being explored by ecologists and hydrologists focus on understanding factors that control rates or distances of nutrient uptake within and among different systems (Fisher et al. 1998). The net effects of transport and transformation can be expressed in terms of retention: the difference in total inputs to and outputs from an aquatic ecosystem, such as a reservoir or a specified length of a river. Because retention integrates physical, chemical, and biological processes occurring throughout an area of interest, and because managers and researchers are trying to determine how to enhance the retentive abilities of streams and rivers (NRC 2000, Mitsch et al. 2001), we will focus on nutrient retention for our consideration of the effects of dam removal on ecosystem dynamics.

Intensive studies of the biogeochemistry of streams and rivers over the past decade have emphasized the importance of transient storage zones for retention of dissolved nutrients—that is, those places in the channel where the flow of water is slowed, allowing sufficient time or circumstances for nutrient processing. Streambeds formed in extensive alluvial deposits or channels with abundant pools and backwaters typically have large amounts of transient storage, and thus have great potential for nutrient retention. Similarly, the size of a channel has an important influence on N processing in streams and rivers; larger channels appear to have an extremely limited ability to influence nutrient loads because of the restricted extent of sediment–water contact relative to the large volumes of water being conveyed (Alexander et al. 2000). In short, the physical structure of the channel can exert an important control on the amount and form of nutrients exported by the stream (D'Angelo et al. 1993, Valett et al. 1996).

Effects of reservoirs on riverine nutrient dynamics

Retention by large reservoir systems can substantially reduce regional nutrient export by rivers (Caraco and Cole 1999), such that the structure and function of receiving coastal systems are fundamentally altered following dam closure (Humborg et al. 1997). Dominant mechanisms of retention are denitrification for N (Jossette et al. 1999) and particle settling for P (Kennedy and Walker 1990). However, it is not clear how these trends of nutrient retention for large reservoirs trans-

late to the smaller impoundments that represent the vast majority of recent removals in the United States. Ecological research on smaller reservoir systems has tended to focus on the role of these structures as barriers to migratory taxa or in creating lentic habitat for other aquatic species (Watters 1996, Benstead et al. 1999), and information regarding their influence on nutrient dynamics is scarce. Even basic questions such as “How many small dams are there?” are unanswered (Poff and Hart 2002). Often, small dams and their impoundments are not included in watershed studies, or it is assumed that these structures have negligible or limited local effects (Graf 1999). Unfortunately, this assumption is largely untested in the context of processes such as primary productivity or nutrient retention. Our limited knowledge of the starting point for subsequent changes represents a distinct challenge for studying the effects of dam removal. Perhaps one of the benefits of the current interest in dam removal will be to enhance our understanding of both the local and cumulative impacts of small impoundments on the dynamics of lotic ecosystems.

Dams do not have to create large impoundments with prolonged hydrologic residence times to foster nutrient retention. Although water may pass through small impoundments quickly relative to large reservoirs, hydrologic residence time is prolonged relative to an unimpounded channel because streamflow has the opportunity to spread out across a wide area. In older impoundments, a long history of sediment trapping means that many of these systems are now quite shallow and may have wetland-like habitats in the upper ends of the reservoir because of bedload deposition and delta formation (figure 1). These broad, shallow channels with reduced water velocities foster sediment deposition, promote P retention, and create greater sediment–water contact needed for denitrification (Kelly et al. 1987, Jansson et al. 1994). In essence, small impoundments represent unusually large transient storage zones in rivers, and the combination of sediment deposition and the creation of wetland or sandbar habitats should promote both P and N retention. Therefore, it is not surprising that we have found that nutrient concentrations immediately below even relatively small impoundments (those with dam heights < 4.5 m) are often less than concentrations upstream of the impoundment (figure 2). The percent reduction between upstream and downstream concentrations that can be achieved by passage through a small reservoir is highly variable. For relatively nutrient-poor systems, this reduction may exceed 70%; for some nutrient-replete systems, the reduction may be as small as 2%.

Channel form and dam removal

If P dynamics are governed by sediment storage and movement, and N retention is determined by the extent of sediment–water interactions, then it may be possible to make general predictions regarding these two critical nutrients from an understanding of changes in channel form and sediment transport triggered by the removal of a dam. Although models of geomorphic changes caused explicitly by dam removal

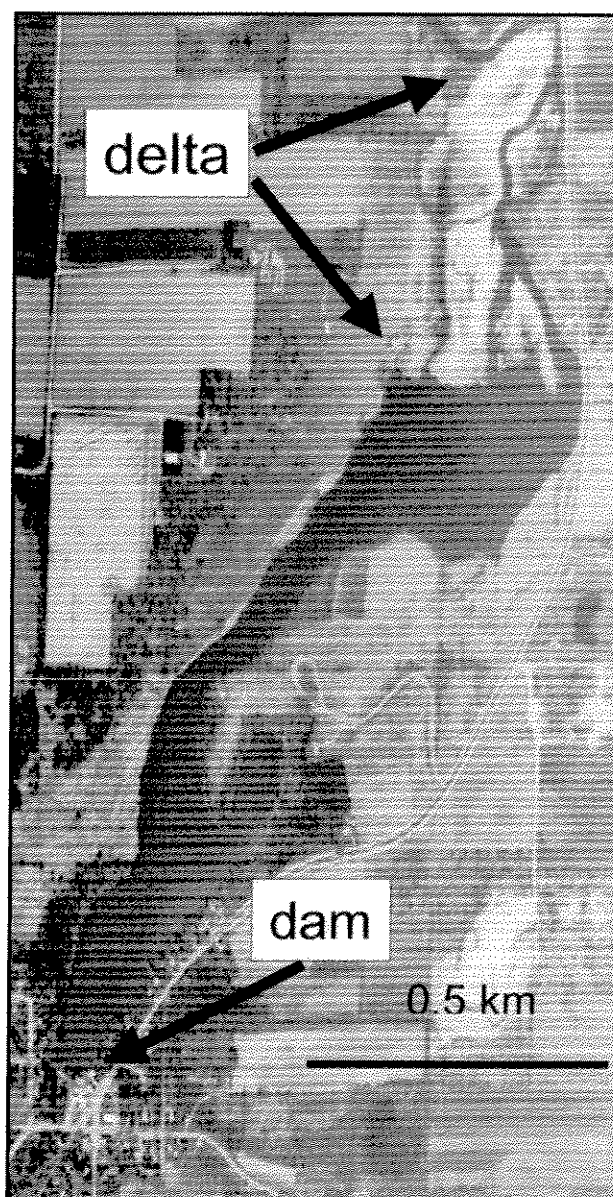


Figure 1. Aerial photo of Rockdale Millpond on Koshkonong Creek, Wisconsin, in 1990. Deposition of sediment at the upstream end of the reservoir has resulted in the formation of an extensive wetland-like delta area. Photograph: US Department of Agriculture, Soil Conservation Society. The photograph is part of the University of Wisconsin's Arthur H. Robinson Map Library collection.

have yet to be developed, there is a wealth of information on how channel form responds to a sudden increase in the slope of the channel, conditions that are frequently created by dam removal in relatively old reservoirs receiving inputs of fine-grained sediment, which are common throughout the midwestern United States.

If the slope of a river channel suddenly increases, for example, by channelization or closure of a meander cutoff, natural processes will act to reestablish equilibrium conditions. Channel adjustment includes a suite of alterations in the

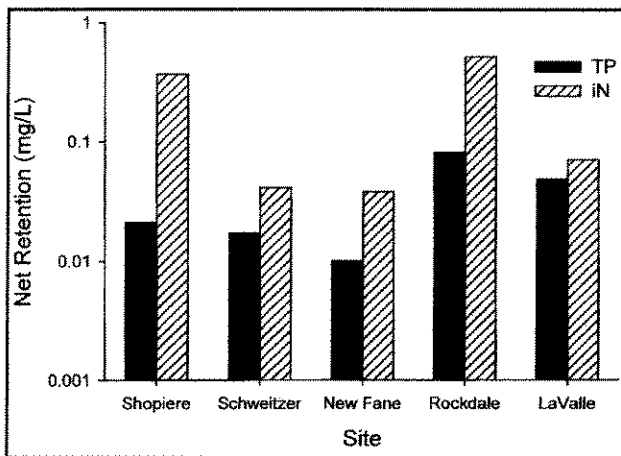


Figure 2. Net retention of total phosphorus (TP) and inorganic nitrogen (iN), determined as the difference in the concentration of water entering versus leaving an impoundment created by low-head, run-of-river dams in southern Wisconsin. Inorganic N = ammonium ($\text{NH}_4\text{-N}$) + nitrate ($\text{NO}_3\text{-N}$) concentration. Values are based on single-day determinations of concentrations above and below impoundments.

width, depth, and alignment of the channel as the system moves back toward equilibrium. These changes have been well documented and synthesized in the form of channel evolution models (Schumm et al. 1984, Simon and Hupp 1986). When dam removal causes an instantaneous increase in slope, these channel evolution models can be used to predict the geomorphic changes in channel form caused by dam removal (Doyle et al. 2002, Pizzuto 2002). It should be emphasized, however, that the specific changes caused by dam removal will vary among different fluvial systems and may include changes other than, or in addition to, an increase in the channel slope, and geomorphic models appropriate to studying dam removal will vary accordingly.

Drawing from the channel evolution model proposed by Simon and Hupp (1986) and from observations from several small dam removals in southern Wisconsin, Doyle and colleagues (2002) suggest that six geomorphic stages of channel development can be recognized within the impounded river reach following the removal of a dam (summarized in figure 3). Herein, we limit the scope of consideration to geomorphic changes within the former impoundment and to the channel only (i.e., we do not consider floodplain devel-

opment). Stage A represents the preremoval, backwatered reservoir condition, which, as described above, often has a broad and relatively shallow form because of sediment trapping within the reservoir. The original channel is often filled completely by sedimentation, leaving little trace of the channel alignment before impoundment. Sediment trapping also means that sediment deposits are often extensive, in some cases filling the entire reservoir (Palmieri et al. 2001).

The most immediate and conspicuous change following breaching of a dam is the rapid decline in the water surface elevation (stage B). The decrease in water depth effectively increases the amount of sediment-water contact before any physical changes to the channel have occurred. The change in the slope of the water surface alone can cause an increase in water velocity, but only in the immediate vicinity upstream of the dam site. In the case of sediment-filled reservoirs, breaching also can cause a rapid increase in the channel slope, which, along with greater water-flow velocity upstream of the dam, initiates the first stage of adjustment to the channel itself (stage C). As the water surface is dropped, the channel degrades vertically into the sediments at the downstream terminus of the former reservoir to create an incised channel, and in the process, large amounts of sediment are transported downstream. Incision appears to begin immediately

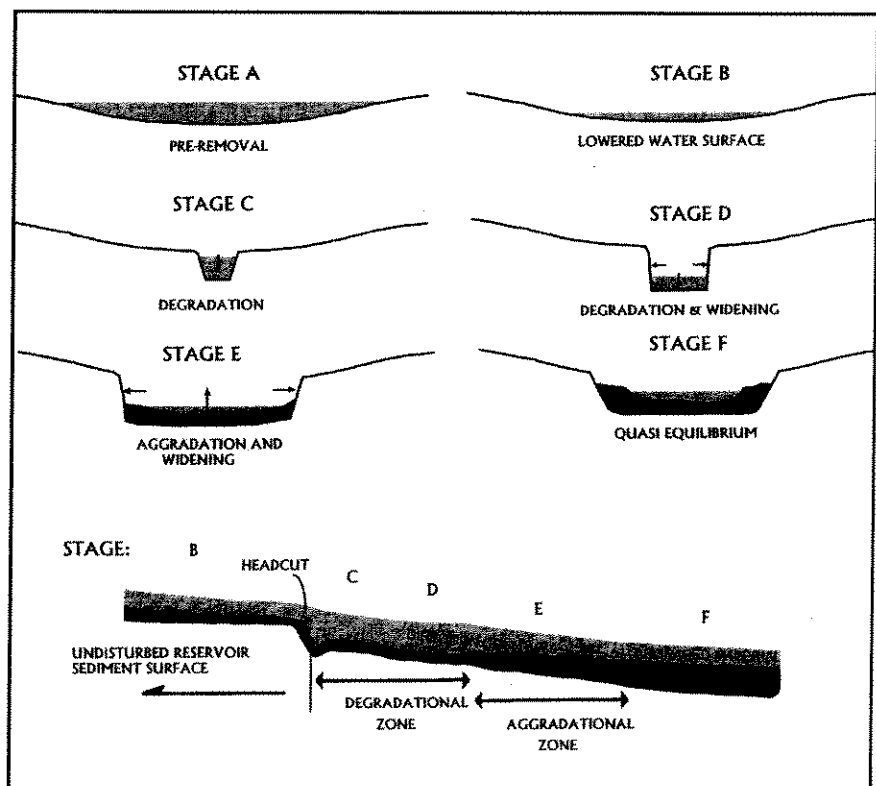


Figure 3. Channel evolution model of geomorphic adjustments following removal of a low-head dam (from Doyle et al. 2002). The upper portion of the figure illustrates changes in the channel cross section that occur at a given place in the channel through time. The lower portion of the figure describes the longitudinal channel profile at a fixed point in time. Figure courtesy of the American Water Resources Association.

following breaching, so we expect that some length of stage C channel will be present along with stage B as soon as the dam is removed.

In rivers dominated by fine-grained sediments, incision creates banks that are often over-steepened, making the deep, narrow channel unstable and prone to slumping. Stage D is characterized by widening of the channel via mass wasting (slumping) of the banks. Reaches undergoing widening often experience substantial sediment losses; in fact, the amount of material lost because of widening can greatly exceed the volume removed by incision during stage C (Grissinger and Murphey 1986). As degradation and widening progress, the sediment derived from upstream erosion is transported to downstream reaches within the former reservoir (and to reaches below the reservoir), where it begins to deposit. The transition between degradation and aggradation within the channel marks the start of stage E. Floodplains begin to form during this stage through overbank deposition as well (Pizuto 2002). Finally, channel form adjustments come to an apparent steady state in stage F with establishment of woody riparian vegetation, thereby stabilizing the channel form.

Following dam removal, the entire reservoir reach does not adjust to the slope change in a uniform fashion, either spatially or temporally. In systems with fine cohesive sediment, channel evolution often begins with the formation of an abrupt vertical drop in the channel slope, known as a headcut or knickpoint (figure 3, longitudinal profile), which subsequently migrates upstream (Schumm et al. 1984). The rate of headcut migration often controls the rate of overall channel adjustment (Ritter et al. 1999). The channel immediately upstream from the headcut experiences little or no alteration other than dewatering (stage B), whereas the downstream reach is fundamentally altered by bed and bank adjustments (stages C and D). The transition between these two stages may be dramatic. For example, following the removal of the Rockdale Dam from Koshkonong Creek, Wisconsin, the river above the headcut was broad and shallow, and no physical changes in the channel had occurred, but below the headcut, water moved rapidly through a narrow, steepened channel (figure 4). Thus following dam removal, the reservoir reach becomes a shifting mosaic of channel forms. With increasing time since removal, the headcut moves farther upstream; more evolutionary stages are likely to be present and more of the entire reservoir will experience some adjustment (figure 5). The rate and extent of adjustment are influenced by site-specific conditions such as the composition of bed and bank material, the cohesion and consolidation of reservoir sediment, or the establishment of vegetation (Thorne 1989, Simon and Rinaldi 2000). However, despite variation in the rate of channel evolution from one dam removal site to another, we expect that the sequence of adjustment is common in many Midwestern rivers similar to Koshkonong Creek. Thus the channel evolution approach provides a valuable framework for understanding physical and, as we outline below, chemical changes following the removal of a dam.

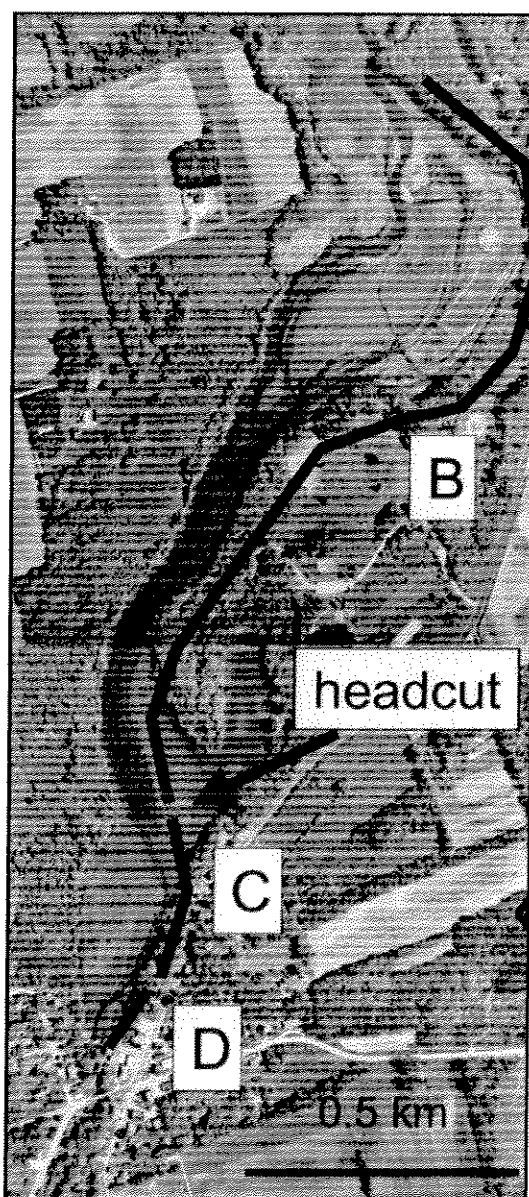


Figure 4. The Rockdale Millpond in May 2001, 7 months after the removal of the dam. Location of the headcut marks the transition between the unadjusted channel (stage B) and reaches experiencing channel evolution. Approximate extents of different channel evolution stages depicted in figure 3 are indicated on the right. Photograph: Wisconsin Department of Transportation.

Nutrient dynamics and channel evolution

Patterns of geomorphic adjustments described above can be summarized in terms of the changes relevant to N and P retention. For N, channel form should be viewed in terms that reflect potential alterations in the degree of sediment–water contact. These changes are, at least in part, captured by measures of the wetted perimeter of the channel over time. For P, the appropriate physical variable is sediment transport.

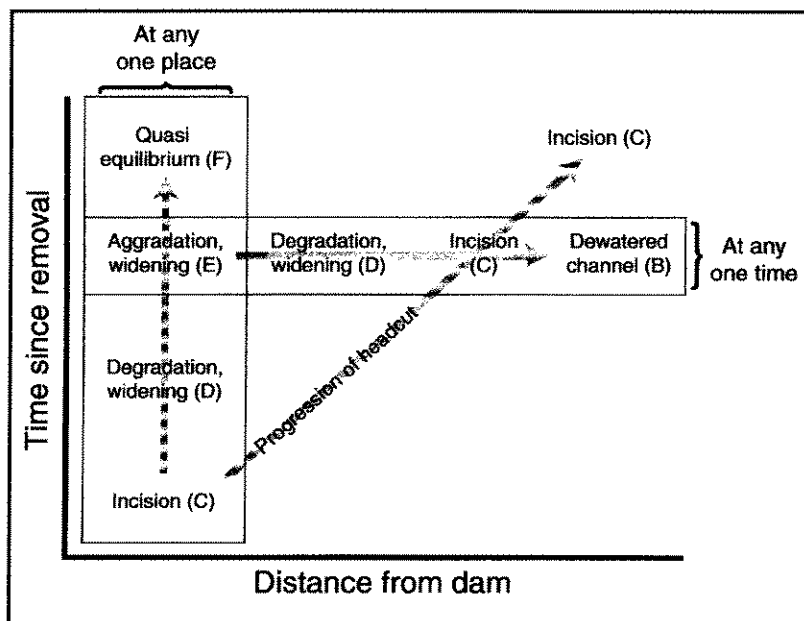


Figure 5. Spatial and temporal distribution of channel evolutionary stages following dam removal. The upstream progression of the headcut marks the transition between the reservoir channel (stage B) and channels undergoing active geomorphic adjustment. As time since removal increases, any one place in the reservoir will progress through the different stages of adjustment; and at any one time, multiple channel stages will be present, with the most rapid progression toward the equilibrium channel occurring in reaches closest to the former dam site. Multiple channel evolution stages present at any one time can be seen in figure 4, in which stages B, C, and D were present 7 months after dam removal.

Before removal (stage A), the wide reservoir area and modest depth characteristic of many small impoundments mean that the wetted perimeter can be an order of magnitude (or more) greater than that of the upstream or downstream channel. Further, many of these sites still retain sediment (i.e., net sediment transport is negative) before removal (figure 6). During stage B, the decline in the water elevation and subsequent dewatering cause only a minor decrease in the wetted perimeter, although the extent of change will depend on the morphometry of the specific reservoir. But as the wetted perimeter decreases, a greater proportion of the water is in contact with the sediment because of the overall reduction in water volume. Because the channel slope has not been altered, slow water velocity persists in the stage B reach, allowing continued sedimentation. Incision during stage C and widening in stage D result in large amounts of sediment transport. The wetted perimeter of the narrow, deep stage C channel is extremely small, but widening in stages D and E steadily increases the wetted perimeter. Aggradation during stage E signals a decrease in sediment transport, although the balance between retention and export of sediment will depend on relative quantities of bank erosion versus bed aggradation. As the channel moves toward a steady-state condition (reduced sediment transport and inputs generally equaling outputs), sed-

iment retention approaches zero while the wetted perimeter gradually increases (figure 6). The extent of the wetted perimeter during stage F could eventually exceed that of the preremoval channel (stage A) if complex channel forms that include features such as backwater areas and side channels are allowed to develop.

Using the logic that P dynamics are driven by sediment transport and N by the extent of sediment-water contact, predictions about these nutrients can be generated from geomorphic trends described by the channel evolution model (figures 3, 6). Greatest P loss should occur from stage C and D channels because sediment transport is maximized, whereas stage E channels should begin to retain P because of aggradation. Similarly, extensive contact between water and sediment enhances N retention during stages A and B, but the reduced contact because of a smaller wetted perimeter, and also to greater water velocities, between nitrate-rich water and the channel in stage C suggests that N retention will be minimal. As the channel widens, sediment-water contact, and thus N retention, are expected to increase progressively in stage D, E, and F channels (figure 6).

Although the generation of predictions regarding transport or retention of different channel stages appears to be relatively straightforward thus far, these stage-based scenarios alone do not resolve the ecosystem-level effects of dam removal on nutrient retention. The amount of N or P retention occurring at any one time following

dam removal will be the product of the types of channel stages present, the spatial extent of each of these stages, and the magnitude of influence of each stage on N and P dynamics. This balancing act can be illustrated by considering geomorphic adjustments observed at Koshkonong Creek following dam removal. Seven months after breaching of the dam, channel stages B, C, and D were recognizable in the former impoundment (figure 4). On the basis of our predictions above, we expect that the extensive area above the headcut in stage B should retain N, whereas little or only modest N retention would occur downstream in reaches that are undergoing active channel evolution (stage C and D channel areas). Cumulatively, this suggests that at this time, the entire reach is likely to be retaining N because of the enhanced retention and large spatial extent of the stage B channel. In contrast, modest sediment and P retention of the stage B channel is likely to be overwhelmed by losses associated with incision and widening below the knickpoint, resulting in a net loss of P from the former reservoir.

We can expand the temporally limited analysis above to suggest some general trajectories for N and P retention following dam removal for the Koshkonong Creek example. Over time, the progression of the headcut and subsequent mass wasting causes increasing export of P because of the mobi-

lization of sediment, as well as the reduction in the extent of the channel area capable of retaining the P entering the system from upstream. Thus P losses from reservoirs may persist and even intensify over time until substantial channel lengths begin to enter into the aggradation stage (stage E). However, N dynamics over the course of dam removal are expected to be distinct from P dynamics. The upstream migration of the headcut marks the transition between two channel stages, one of which should retain N (stage B), perhaps even more strongly than the preremoval channel, and the other of which is likely to do little more than transport N downstream (stage C). Thus, on the basis of our observations of channel adjustment, it may be several months before measurable declines in N retention associated with shrinkage of the B channel are detectable at this site.

Assumptions and implications of linking geomorphic and ecological models

In considering this geomorphic framework for understanding changing nutrient retention following dam removal, we have made some assumptions and simplifications that need to be addressed. The most important assumption is that NO_3^- and particulate P dominate the total N and P budgets, respectively. The assumption of NO_3^- and particulate P dominance led us to a second assumption, that N retention is most strongly influenced by denitrification and P retention is driven by either the settling or transport of sediments. For N, the assumptions that NO_3^- represents the majority of N in transport and, more important, that denitrification is the dominant mechanism of retention, appear to be reasonable for even slightly enriched systems (Hedin et al. 1995, Saunders and Kalff 2001). However, we have not considered a potentially important pool of N in the form of particulate N, which includes both organic particles and ammonium (NH_4^+) sorbed to sediments. While sedimentation of particulate N does not make a large contribution to N retention in lakes and wetlands (Saunders and Kalff 2001), NH_4^+ concentrations in reservoir sediments can be extremely high. It is reasonable to assume that mobilization of sediments associated with incision and widening will promote N export from the reservoir in a fashion similar to P (Perrin et al. 2000). For P, water-soluble forms of this nutrient often represent a substantial fraction of total P load. Nonetheless, mobilization and transport of sediment is still likely to exert an important influence on P retention following dam removal because dissolved P (notably, phosphate) will sorb to sediment particles that become entrained in the water column during channel adjustment.

An obvious simplification in this analysis has been to restrict our consideration of nutrient dynamics only to the section of river affected by impoundment. However, the ef-

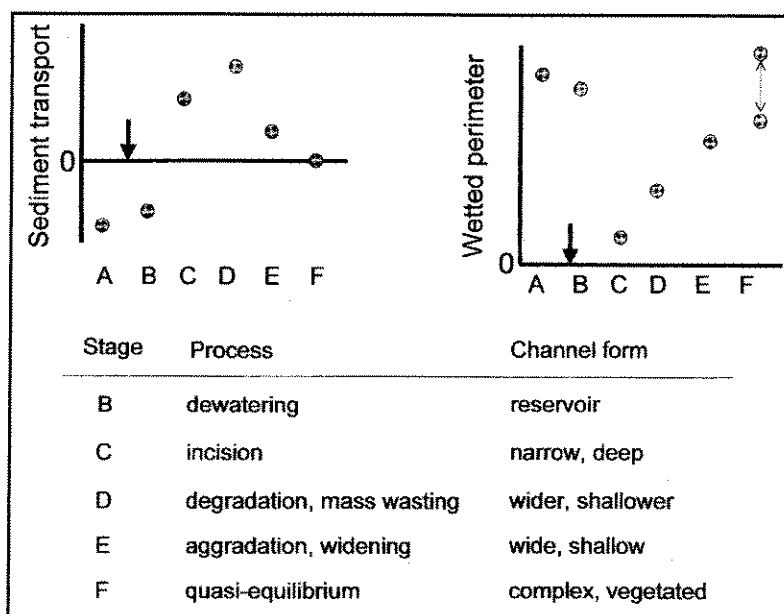


Figure 6. Summary of the relative amounts of sediment export and wetted perimeter, and the characteristic processes and channel forms for the geomorphic stages of channel evolution following dam removal. Positive values for sediment transport indicate a net loss of sediment from the reservoir reach; negative values denote retention of sediments within the reservoir reach. Arrows indicate timing of dam removal.

fects of dams and dam removal are best understood in the larger context of the watershed. Although changes within the impounded reach are rapid and dramatic following dam removal, effects of removal may be measurable for several kilometers downstream. Depending on particle size, downstream deposition of reservoir sediments might either increase or decrease transient storage and, therefore, nutrient uptake below the dam site. Fine-particle deposition can potentially clog interstitial spaces and reduce the movement of water into and out of the streambed, or alternatively, deposition of coarse particles can lead to the formation or enhancement of bars or other bedform features, increasing sediment-water contact (Stanley et al. 2002). The balance between within-reservoir versus downstream effects of dam removal remains an important area of investigation.

A further simplification of our conceptual framework is the use of a single geomorphic parameter for assessing N dynamics, although other aspects of channel morphology and hydraulics undoubtedly play important roles. For simplicity and clarity, we have used only channel wetted perimeter, although a measure of the proportion of flow in contact with the bed would be more desirable. Also, transitions between different channel stages will affect water velocity through the channel, which in turn plays an important role in determining nutrient uptake rates (Wolheim et al. 2001). By using a single parameter to characterize the physical changes occurring within the river, we have not distinguished between the interrelated effects of changing water velocity and chan-

nel form. We expect that changes in velocity caused by altered channel form should intensify predicted nutrient responses. For example, the large amount of sediment–water contact predicted from the extensive wetted perimeter of the stage B channel will be enhanced by the slow flow rate of water over this large area. In contrast, the small wetted perimeter or extent of sediment–water contact in stage C is further reduced by high water velocity.

Despite these caveats, several points regarding the effects of dam removal emerge from this analysis. The major theme emphasized here is that changes in nutrient retention caused by dam removal are expected to be shaped by geomorphic channel adjustments. Although the specific way in which channels adjust to dam removal will vary from region to region, the change in the physical template will strongly influence the ecological responses to dam removal. Changes in nutrient retention following dam removal should be complex, reflecting a balance between the dynamics of channel adjustment and the relative influence of different channel stages on N and P processing. Following dam removal, affected sections of a river may consist of a series of reaches that have distinct and potentially contrasting influences on the form and amount of nutrients being transported downstream.

Dam removal represents an extreme example of the influence of channel morphology on nutrient dynamics in streams and rivers. While it is impossible to ignore the dramatic and relatively rapid geomorphic changes when studying dam removal, streams and rivers are dynamic physical systems subject to short- and long-term changes in channel form. Yet channel geomorphology is usually treated as a fixed or constant attribute in ecological studies. As channels change over time or from site to site in a stream, the extent of sediment–water contact or the rate of sediment transport will also vary. For example, the transition from a narrow and deep incised channel to a broader, shallow channel under conditions of constant discharge will be accompanied by a decline in water velocity, an increase in wetted perimeter, and probably an increase in the amount of interstitial flow. Because geomorphic adjustments can alter sediment transport and the extent of sediment–water contact, changing channel form alone has the potential to affect uptake lengths and rates of biologically important elements such as N and P. Although the general importance of channel form on a range of ecosystem processes, including nutrient cycling, is well established (Brussock et al. 1985, Frissell et al. 1986, D'Angelo et al. 1997), we are only beginning to understand how specific geomorphic attributes constrain nutrient dynamics in lotic systems. Dam removal can be used as an experiment for testing predictions or quantifying relationships between the dynamics of channel form and nutrient retention and thus represents a rare opportunity to gain valuable insights into the transport and transformation of nutrients as they move through watersheds.

Acknowledgments

We are grateful to David Hart and LeRoy Poff for organizing this special issue on dam removal, and for providing us with valuable suggestions for this article. Karen Bushaw-Newton, Jim Pizzuto, and an anonymous reviewer also provided many thoughtful comments that greatly improved this article. Michael Barry assisted with graphics. This research was supported by the Purdue University Showalter Fund, the Bradley Fund for the Environment, a Horton Grant to M. W. D. from the American Geophysical Union, and NSF grant DEB-0108619 to E. H. S. We are grateful for the support The Pew Charitable Trusts provided, which allowed us to complete this article.

References cited

- Alexander RB, Smith R, Schwarz GE. 2000. Effect of stream channel size on the delivery of nitrogen to the Gulf of Mexico. *Nature* 403: 758–761.
- Bennett EM, Carpenter SR, Caraco NF. 2001. Human impact on erodable phosphorus and eutrophication: A global perspective. *BioScience* 51: 227–234.
- Benstead JP, March JG, Pringle CM, Scatena FN. 1999. Effects of a low-head dam and water abstraction on migratory tropical stream biota. *Ecological Applications* 9: 656–668.
- Born SM, Genskow KD, Filbert TL, Hernandez-Mora N, Keefer ML, White KA. 1998. Socioeconomic and institutional dimensions of dam removals: The Wisconsin experience. *Environmental Management* 22: 359–370.
- Brussock PP, Brown AV, Dixon JC. 1985. Channel form and stream ecosystem models. *Water Resources Bulletin* 21: 859–866.
- Caraco NF, Cole JJ. 1999. Human impact on nitrate export: An analysis using major world rivers. *Ambio* 28: 167–170.
- Carpenter SR, Caraco NF, Correll DL, Howarth RW, Sharpley AN, Smith VH. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecological Applications* 8: 559–568.
- D'Angelo DJ, Webster JR, Gregory SV, Meyer JL. 1993. Transient storage in Appalachian and Cascade mountain streams as related to hydraulic characteristics. *Journal of the North American Benthological Society* 12: 223–235.
- D'Angelo DJ, Gregory SV, Ashkenas LR, Meyer JL. 1997. Physical and biological linkages within a stream geomorphic hierarchy: A modeling approach. *Journal of the North American Benthological Society* 16: 480–502.
- Doyle MW, Stanley EH, Luebke MA, Harbor JM. 2000. Dam removal: Physical, biological, and societal considerations. Paper presented at the American Society of Civil Engineering Joint Conference on Water Resources Engineering and Water Resources Planning and Management; 30 July–2 August; Minneapolis, MN.
- Doyle MW, Stanley EH, Harbor JM. 2002. Assessing probable channel response to dam removal using geomorphic analogies. *Journal of the American Water Resources Association*. Forthcoming.
- Fisher SG, Grimm NB, Marti E, Holmes RM, Jones JB. 1998. Material spiraling in stream corridors: A telescoping ecosystem model. *Ecosystems* 1: 19–34.
- Frissell CA, Liss WJ, Warren CE, Hurley MC. 1986. A hierarchical framework for stream habitat classification: Viewing streams in a watershed context. *Environmental Management* 10: 99–214.
- García-Ruiz R, Pattinson SN, Whitton BA. 1998. Denitrification in river sediments: Relationship between process rate and properties of water and sediment. *Freshwater Biology* 39: 467–476.
- Goolsby DA, Battaglin WA, Lawrence GB, Artz RS, Aulenbach BT, Hooper RP, Keeney DR, Stensland GJ. 1999. Flux and Sources of Nutrients in the Mississippi-Atchafalaya River Basin: Topic 3 Report for the Integrated Assessment on Hypoxia in the Gulf of Mexico. Washington (DC): White House Office of Science and Technology Policy, Committee on Environment and Natural Resources, Hypoxia Work Group. (3 July 2002; <http://wwwrcolka.cr.usgs.gov/midconherb/final.pdf>)

- Graf WL. 1999. Dam nation: A geographic census of American dams and their large-scale hydrologic impacts. *Water Resources Research* 35: 1305–1311.
- Grissinger EH, Murphey JB. 1986. Bank and bed adjustments in a Yazoo Bluffline tributary. Pages 1003–1012 in Wang SSS, Shen HW, Ding LZ, eds. *Third International Symposium on River Sedimentation*. Oxford: University of Mississippi Press.
- Hedin LO, Armesto JJ, Johnson AH. 1995. Patterns of nutrient loss from unpolluted, old-growth temperate forests: Evaluation of biogeochemical theory. *Ecology* 76: 493–509.
- Humborg C, Ittekkot V, Cociasu A, Bodungen BV. 1997. Effect of Danube River dam on Black Sea biogeochemistry and ecosystem structure. *Nature* 386: 385–388.
- Jansson M, Andersson R, Berggren H, Leonardson L. 1994. Wetlands and lakes as nitrogen traps. *Ambio* 23: 320–325.
- Josette G, LePorcq B, Sanchez N, Philippon P. 1999. Biogeochemical mass-balances (C, N, P, Si) in three large reservoirs of the Seine basin (France). *Biogeochemistry* 47: 119–146.
- Kelly CA, Rudd JWM, Hesslein RH, Schindler DW, Dillon PJ, Driscoll CT, Gherini SA, Hecky RE. 1987. Prediction of biological acid neutralization in acid-sensitive lakes. *Biogeochemistry* 3: 129–141.
- Kennedy RH, Walker WW. 1990. Reservoir nutrient dynamics. Pages 109–132 in Thornton KW, Kimmel BL, Payne FE, eds. *Reservoir Limnology: Ecological Perspectives*. New York: Wiley-Interscience.
- Martini EC. 1998. Wisconsin's Milldam Act: Drawing new lessons from an old law. *Wisconsin Law Review* 1998: 1305–1336.
- Mitsch WJ, Day JW, Gilliam JW, Groffman PM, Hey DL, Randall GW, Wang N. 2001. Reducing nitrogen loading to the Gulf of Mexico from the Mississippi River basin: Strategies to counter a persistent ecological problem. *BioScience* 51: 373–388.
- Newbold JD, Elwood JW, O'Neill RV, Van Winkle W. 1981. Measuring nutrient spiraling in streams. *Canadian Journal of Fisheries and Aquatic Sciences* 38: 860–863.
- Ng H, Mayer T, Marsalek J. 1993. Phosphorus transport in runoff from a small agricultural watershed. *Water Science and Technology* 28: 451–460.
- [NRC] National Research Council. 1992. *Restoration of Aquatic Ecosystems*. Washington (DC): National Academy Press.
- . 2000. *Clean Coastal Waters: Understanding and Reducing the Effects of Nutrient Pollution*. Washington (DC): National Academy Press.
- Palmieri A, Shah F, Dinar A. 2001. Economics of reservoir sedimentation and sustainable management of dams. *Journal of Environmental Management* 61: 149–163.
- Perrin CJ, Ashley KI, Larkin GA. 2000. Effects of drawdown on ammonium and iron concentrations in a coastal mountain reservoir. *Water Quality Research Journal of Canada* 35: 231–244.
- Pizzuto J. 2002. Effects of dam removal on river form and process. *BioScience* 52: 683–691.
- Poff NL, Hart DD. 2002. How dams vary and why it matters for the emerging science of dam removal. *BioScience* 52: 659–668.
- Ritter DF, Kochel RC, Miller JR. 1999. The disruption of Grassy Creek: Implications concerning catastrophic events and thresholds. *Geomorphology* 29: 323–338.
- Royer T, Tank JL, David M. 2001. Is denitrification a major sink for nitrate in agricultural streams? *Bulletin of the North American Benthological Society* 18: 208.
- Saunders DL, Kalf J. 2001. Nitrogen retention in wetlands, lakes, and rivers. *Hydrobiologia* 443: 205–212.
- Schumm SA, Harvey MD, Watson CC. 1984. *Incised Channels: Morphology, Dynamics and Control*. Littleton (CO): Water Resources Publications.
- Simon A, Hupp CR. 1986. Channel evolution in modified Tennessee channels. Pages 71–82 in Section 2, *Proceedings of the Fourth Federal Interagency Sedimentation Conference, Las Vegas, NV*. Washington (DC): US Interagency Advisory Committee on Water Data.
- Simon A, Rinaldi M. 2000. Channel instability in the Loess Area of the Midwestern United States. *Journal of the American Water Resources Association* 36: 133–150.
- Stanley EH, Luebke MA, Doyle MW, Marshall DW. 2002. Short-term changes in channel form and macroinvertebrate communities following low-head dam removal. *Journal of the North American Benthological Society* 21: 172–187.
- Thorne CR. 1989. Effects of vegetation on riverbank erosion in addition to stability. Pages 125–144 in Thornes JB, ed. *Vegetation and Erosion*. Chichester (UK): John Wiley and Sons.
- [USACE] US Army Corps of Engineers. 1998. *National Inventory of Dams*. {3 July 2002; <http://crunch.tec.army.mil/nid/webpages/nid.cfm>}
- Valett HM, Morrice JA, Dahm CN, Campana ME. 1996. Parent lithology, surface-groundwater exchange, and nitrate retention in headwater streams. *Limnology and Oceanography* 41: 333–345.
- Watters GT. 1996. Small dams as barriers to freshwater mussels (*Bivalvia*, *Unionida*) and their hosts. *Biological Conservation* 75: 79–85.
- Webster JR, Patten BC. 1979. Effects of watershed perturbation on stream potassium and calcium dynamics. *Ecological Monographs* 49: 51–72.
- Wolheim WM, Peterson BJ, Deegan LA, Hobbie JE, Hooker E, Bowden WB, Edwardson KJ, Arscott DB, Hershey AE, Finlay JC. 2001. Influence of stream size on ammonium and suspended particulate nitrogen processing. *Limnology and Oceanography* 46: 1–13.
- [WDNR] Wisconsin Department of Natural Resources. 1995. *Wisconsin's biodiversity as a management issue: A report to the Department of Natural Resources Managers*. Madison (WI): WDNR.

Dam Removal: Challenges and Opportunities for Ecological Research and River Restoration

DAVID D. HART, THOMAS E. JOHNSON, KAREN L. BUSHAW-NEWTON, RICHARD J. HORWITZ, ANGELA T. BEDNAREK, DONALD F. CHARLES, DANIEL A. KREEGER, AND DAVID J. VELINSKY

Water flow is a “master variable” (*sensu* Power et al. 1995) that governs the fundamental nature of streams and rivers (Poff et al. 1997, Hart and Finelli 1999), so it should come as no surprise that the modification of flow caused by dams alters the structure and function of river ecosystems. Much has been learned during the last several decades about the adverse effects of dams on the physical, chemical, and biological characteristics of rivers (Ward and Stanford 1979, Petts 1984, Poff et al. 1997, Poff and Hart 2002). Increasing concerns about these impacts, together with related social and economic forces, have led to a growing call for the restoration of rivers by removing dams (AR/FE/TU 1999, Pejchar and Warner 2001). For the purposes of this paper, we define restoration broadly as an effort to compensate for the negative effects of human activities on ecological systems by facilitating the establishment of natural components and regenerative processes, although we acknowledge that these efforts rarely eliminate all human impacts (see Williams et al. 1997 for alternative definitions).

Interest in dam removal as a means of river restoration has focused attention on important new challenges for watershed management and simultaneously created opportunities for advancing the science of ecology. One challenge lies in determining the magnitude, timing, and range of physical, chemical, and biological responses that can be expected following dam removal. This information is needed to decide whether and how dam removals should be performed to achieve specific restoration objectives (Babbitt 2002). Opportunities for advancing ecological research also exist because dam removal represents a major, but partially controllable, perturbation that can help scientists test and refine models of complex ecosystems. In contrast to the small-scale experiments that traditionally have been employed in stream and river ecology, the unusually large magnitude and spatial extent of dam removal

WE DEVELOP A RISK ASSESSMENT FRAMEWORK FOR UNDERSTANDING HOW POTENTIAL RESPONSES TO DAM REMOVAL VARY WITH DAM AND WATERSHED CHARACTERISTICS, WHICH CAN LEAD TO MORE EFFECTIVE USE OF THIS RESTORATION METHOD

“experiments” creates the potential for examining river responses by means of both mechanistic and whole-system approaches.

David D. Hart (e-mail: hart@acnatsci.org) is an ecologist, academy vice president, and director of the Patrick Center for Environmental Research, Academy of Natural Sciences, 1900 Benjamin Franklin Parkway, Philadelphia, PA 19103. Thomas E. Johnson is a hydrologist and Watershed Hydrology section leader at the Patrick Center for Environmental Research. Karen L. Bushaw-Newton is an aquatic ecologist and postdoctoral fellow at the Patrick Center for Environmental Research; she is currently an assistant professor of biology at American University, Washington, DC. Richard J. Horwitz is an aquatic ecologist and Fisheries section leader at the Patrick Center for Environmental Research. Angela T. Bednarek is a PhD candidate and Morris Udall Fellow in Environmental Policy at the University of Pennsylvania, Philadelphia, PA 19104. Donald F. Charles is an algal ecologist and Phycology section leader at the Patrick Center for Environmental Research. Daniel A. Kreeger is a physiological ecologist and Wetland Ecology section leader at the Patrick Center for Environmental Research. David J. Velinsky is an environmental geochemist and Environmental Geochemistry section leader at the Patrick Center for Environmental Research. © 2002 American Institute of Biological Sciences.

The overall objectives of this article are to assess the current understanding of ecological responses to dam removal and to develop a new approach for predicting dam removal outcomes based on stressor–response relationships. We begin by explaining how a simplified spatial and temporal context can be helpful for examining dam removal responses. Three alternative approaches for predicting ecological responses to dam removal are then evaluated: (1) predictions based on studies of actual dam removals; (2) predictions based on studies of existing dams; and (3) predictions based on mechanistic and empirical models (e.g., sediment transport models).

A preliminary conclusion of this evaluation is that useful generalizations about dam impacts and ecological responses to dam removal cannot be made without considering the nature of stress imposed by dams of different size and operational type across a variety of watershed settings. Furthermore, expected responses to removal are often based on knowledge about large (e.g., > 15 meters [m] height) flood control or hydropower dams that can dramatically alter water quality and flow regimes (Petts 1984), whereas most of the dams being removed are relatively small structures (≤ 5 m height) that may have less marked effects on river ecosystems. There is relatively little information on the ecological impacts of these smaller dams, however, and the limited studies of small dam removals have yielded variable results. To address this knowledge gap, we develop a risk assessment framework for evaluating relationships between dam impacts and dam characteristics across a broad range of dam sizes, with the ultimate goal of predicting restoration outcomes for different types of dams and rivers.

Finally, we briefly explore two additional issues associated with the use of dam removal in watershed management. First, although the long-term ecological benefits of dam removal are potentially quite large, the removal process can also have some adverse effects on river ecosystems. Thus, there is a need to develop methods for anticipating and mitigating these impacts. Second, dam removal is but one of many potential tools and practices for restoring and protecting rivers, so comprehensive approaches are required to determine the best combination of methods for achieving watershed management goals.

A spatial and temporal context for examining ecological responses to dam removal

Efforts to understand dam removal responses must first consider how these responses are likely to vary in space and time (figure 1). Responses to dam removal include those that re-

sult from the removal process itself, as well as changes that occur when various impacts caused by the dam's presence are eliminated. The rate, magnitude, duration, and spatial extent of these changes will depend on various characteristics of the dam, river, and watershed (Poff and Hart 2002), as well as the method of dam removal.

Spatially, it is useful to distinguish among responses to dam removal that occur downstream from the dam, within impounded areas, and in the free-flowing areas farther upstream. For example, when the impoundment becomes free-flowing following dam removal, changes can occur in a variety of important hydraulic parameters (e.g., slope, velocity field) and geomorphic processes (e.g., channel incision, bank failure) that influence habitat conditions. In areas downstream from the dam, the erosion and downstream transport of accumulated sediment from the former impoundment can lead to deposition and other channel changes. Changes in flow regime (including the size, timing, and duration of maximum and minimum flows) in this downstream area can range from minor, in the case of a 2-m-high mill dam, to major, in the case of a 50-m-high peaking hydropower dam or other highly regulated dam. The principal effects of dam removal upstream of the impoundment are likely to be mediated through biotic responses to the restoration of connectivity, including upstream colonization by migratory fauna and associated nutrient transport and genetic changes.

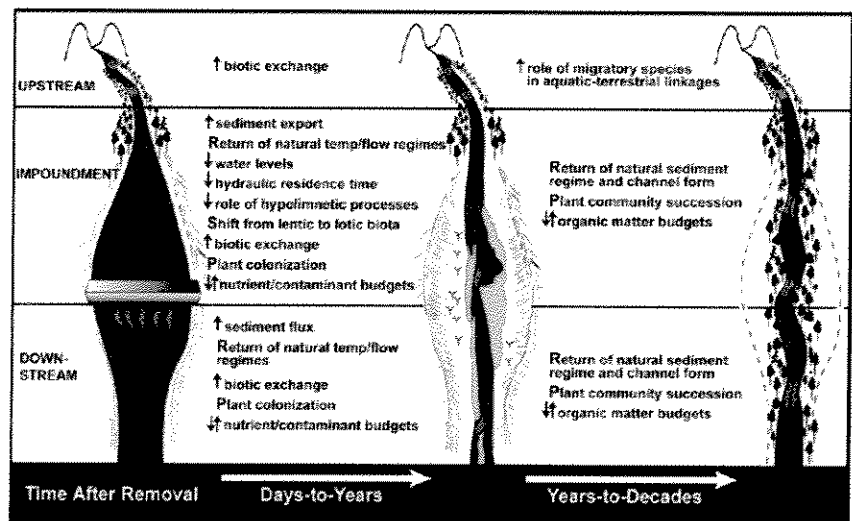


Figure 1. A simple spatial and temporal context for examining potential ecological responses to dam removal. Prior to removal, upstream and downstream free-flowing areas are separated by an impoundment. Dam removal initiates a series of abiotic and biotic changes that vary among areas and occur at different rates. For example, the rate of sediment transport and channel adjustment is a function of the distribution of sediment particle sizes and flow magnitudes, and the response rate of aquatic and riparian biota to these changes depends on their dispersal and growth rates. Key changes occurring within each spatial and temporal area have been highlighted. For some processes, arrows indicate net change as either increases (↑) or decreases (↓), though in other cases the change may be in either direction (↑↓).

There may also be reductions in fauna that formerly dispersed upstream from the impoundment.

Ecological responses to dam removal can also occur over a broad range of time scales. For example, short-term changes associated with the downstream transport of fine sediment from the former impoundment begin as soon as the dam is breached, and fish whose upstream movements were formerly obstructed by the dam may begin to move into the former impoundment within days after removal. Over longer periods, changes in channel morphology generally propagate upstream from the dam site by headward erosion. Establishment of an equilibrium channel morphology, new floodplains, and native riparian vegetation in the former impoundment area may take much longer, on the order of years to decades. Similarly, some faunal changes may occur rapidly (within days), but other long-term changes occur as species adjust to changes in channel form.

Alternative approaches for predicting ecological responses to dam removal

Observed ecological responses to dam removal.

One approach for developing predictive models is by means of the analysis and synthesis of results from a large set of dam removal studies. This approach, however, is currently limited by the scarcity of scientific studies of actual dam removals (Bednarek 2001). Although more than 450 dams have been removed in the United States during the last century (AR/FE/TU 1999), less than 5% (approximately 20) of these removals were accompanied by published ecological studies. We are also aware of about 10 ongoing studies (as of December 2001) that will contribute further to our understanding of ecological responses to dam removal. The knowledge gained from these newer studies, however, is restricted to an understanding of relatively short-term changes. In contrast, recovery of certain ecological attributes may take years to decades. Nonetheless, we can begin to summarize some of the physical, chemical, and biological responses to removal that have been documented to date (table 1).

Shifts in patterns of sediment movement have been one of the most prominent and significant ecological responses to dam removal. Changes in sediment transport control the process of channel evolution (e.g., the rate of headward erosion in the former impoundment, the aggradation of downstream reaches, channel narrowing, creation of new floodplains), which also has important consequences for biogeochemical cycling and habitat availability. Although dam removal allows sediment stored in the impoundment to be transported downstream, observed rates and patterns of sediment transport can be quite variable, depending on the amount and type of sediment, channel slope, and flow magnitude. Many studies refer to increased sediment flux following dam removal (e.g., Clearwater River dams, Shopiere Dam, Woolen Mills Dam; table 1), but few have attempted to quantify sediment transport rates. In the first 9 years after the removal of the Newaygo Dam on the Muskegon River (MI), Si-

mons and Simons (1991) estimated that the downstream rate of sediment movement averaged nearly 2 km per year (median grain size = 0.25 mm). They estimated that complete flushing of the system could take an additional 50 to 80 years (Simons and Simons 1991). Mobilization of fine-grained sediment was also reported immediately following the removals of dams on several other rivers (e.g., the Clearwater, Baraboo, AuSable, Mad, and Milwaukee Rivers; table 1). Accumulated sediment may be coarse grained, however, and not easily mobilized. For example, Johnson and colleagues (2001) observed little increase in suspended or bedload transport during the breaching of the Manatawny Creek Dam (table 1). Rather, most of the sediment (median grain size = 45 mm) did not move downstream until several months later when discharge increased from less than $3 \text{ m}^3 \cdot \text{sec}^{-1}$ to nearly $100 \text{ m}^3 \cdot \text{sec}^{-1}$. No quantitative geomorphic study has continued long enough to document the establishment of an equilibrium channel morphology following dam removal, although the time frame could range from years to decades or more (Pizuto 2002).

Dam removal can affect water quality through the downstream transport of sediment-bound contaminants (e.g., organic substances and heavy metals) and the alteration of biogeochemical cycles. For example, a large volume of fine sediment contaminated with polychlorinated biphenyls (PCBs) was present in the impoundment upstream of Ft. Edward Dam on the Hudson River, and these contaminants were transported downstream when the dam was breached (Shuman 1995). Unfortunately, the dam owner did not perform an adequate preremoval assessment of potential sediment contamination, despite knowledge that PCBs were produced in an upstream industrial facility (Shuman 1995). The impoundment created by a small mill dam on the Manatawny Creek in southeastern Pennsylvania also contained some contaminants within the sediments (e.g., heavy metals, PCBs, and polycyclic aromatic hydrocarbons, or PAHs), but this situation was very different from that in the Hudson River (Bushaw-Newton et al. 2001). Specifically, the fine sediments to which these contaminants preferentially sorb were very uncommon in the impoundment, so the total volume of contaminated sediment was minimal. Moreover, concentrations of these contaminants per unit of fine sediment were generally low, and similar concentrations were observed in fine sediment samples collected from free-flowing reaches located upstream and downstream of the dam. One exception to this pattern occurred for PAHs, which exhibited elevated concentrations at a few locations within the impoundment, presumably because of the dam's urban setting. Sediment contamination has not been a major issue for many other dam removals, however. For example, preremoval studies of Salling Dam on the AuSable River in Michigan indicated that the sediment primarily comprised flocculated organics, and no contaminants were present (Pawloski and Cook 1993). Future efforts to assess the risks associated with potential sediment contamination should focus particular attention on current and former human activities within the watershed, as well as

Table 1. Observed effects of dam removal on the physical, chemical, and biological components of a river ecosystem.

Dam river system (dam life span ^a)	Estimated Size ^b		Physical ^c	Chemical	Biological	Reference
	Height by Length (meters)	Impoundment (hectares)				
Dead Lake Dam Chipola River, FL (1960–1987)	5 x 240 2700		Alteration of flow regime	Improvement in overall water quality	Restoration of fish passage; increase in fish diversity	Estes et al. 1993, Hill et al. 1994
Edwards Dam Kennebec River, ME (1837–1999)	7 x 280 462		Erosion at dam site; bank slumping at deepest section of former impoundment		Shift from pelagic to benthic algae in former impoundment; restoration of fish passage (striped bass and sturgeon); plant colonization	Casper et al. 2001, O'Donnell et al. 2001
Ft. Edward Dam Hudson River, N.Y. (1898–1973)	9 x 179 79		Increased sediment transport	Mobilization of organic contaminants		Shuman 1995
Fulton Dam Yahara River, WI (1849–1993)	3 x n.d. 20				Change in community composition; loss of reservoir species	ASCE 1997, Born et al. 1998
Grangeville Dam Clearwater River, ID (1903–1963)	17 x 134 n.d.		Increased sediment transport			Winter 1990
Jackson Street Dam^d Bear Creek, OR (1960–1998)	3 x 37 1				Restoration of fish passage (salmon)	Smith et al. 2000
Kettle River Dam Kettle River, MN (1915–1995)	6 x 46 n.d.		Increased sediment transport		Decrease in mussel abundance downstream due to sedimentation	Johnson 2001
Lewiston Dam Clearwater River, ID (1927–1973)	14 x 323 n.d.		Increased sediment transport		Restoration of fish passage (salmon); improvement of fish habitat	Williams 1977, Winter 1990, Shuman 1995
Manatawny Creek Dam Manatawny Creek PA (late 1700s–2000)	2 x 30 1.5		Increased sediment; transport; downstream channel aggradation; channel formation; and channel substrate coarsening in former impoundment	Minimal contaminant storage; no change in most forms of nitrogen and phosphorous between upstream and and downstream	Shift in macroinvertebrate and fish species composition from lentic to lotic in former impoundment; decrease in fish parasites in former impoundment; stranding of organisms due to drawdown; plant colonization	Bushaw-Newton et al. 2001, Hart et al. 2001, Horwitz et al. 2001, Johnson et al. 2001
Nelsonville Dam Tomorrow River, WI (1860–1988)	2 x n.d. 9		Decreased water temperatures		Spawning of trout; reclassified as a class 1 trout fishery	Born et al. 1998
Newaygo Dam Muskegon River, MI (1853–1968)	n.d. n.d.		Increased sediment transport			Simons and Simons 1991
Oak Street Dam Baraboo River, WI (1860–2000)	4 x 63 6–15		Increased sediment transport; channel formation		Shift in benthic macroinvertebrate species composition from lentic to lotic in former impoundment; increase in fish community quality in former impoundment; decrease in fish community quality downstream	Catalano et al. 2001, Stanley et al. 2002

Continued from previous page

Dam river system (dam life span ^a)	Estimated size ^b Height by Length (meters) Impoundment (hectares)	Physical ^c	Chemical	Biological	Reference
Quaker Neck Dam Neuse River, N.C. (1952–1998)	2 x 79 n.d.			Restoration of fish passage (American shad and striped bass)	Bowman 2001
Rockdale Dam Koshkonong Cr, WI (1848–2000)	2 x 23 42	Increased sediment transport	Mobilization of phosphorus; predicted loss of nitrogen retention		Stanley and Doyle 2001, 2002
Salling Dam AuSable River, MI (1914–1991)	5 x 76 22	Increased sediment transport; decreased water temperature in former impoundment and downstream	No contaminated sediment	Plant colonization	Pawloski and Cook 1993
Shoplere Dam Turtle Cr, WI (1848–1999)	4 x n.d. 6	Increased sediment transport		Change in benthic macroinvertebrate species composition at former damsite	Pollard and Reed- Anderson 2001
Stronach Dam^d Pine River, MI (1912–1996 ongoing)	4 x 23 12	Progressive downcutting and transport of sediment; Increased water velocity in former impound- ment; downstream channel aggradation		Increase in lotic fish species in former impoundment (brown and rainbow trout)	Burroughs et al. 2001
Sweasey Dam Mad River, CA (1938–1969)	17 x n.d. n.d.	Increased sediment transport		Improved fish passage	Winter 1990
Waterworks Dam Baraboo River, WI (1858–1997)	4 x n.d. 19	Increased sediment transport; channel formation		Shift in macroinvertebrate species composition from lentic to lotic in former impoundment; increase in quality of fish community in former impoundments; decrease in quality of fish community downstream	Catalano et al. 2001, Stanley et al. 2002
Woolen Mills Dam Milwaukee River, WI (1870–1988)	6 x n.d. 27	Increased sediment transport		Increase in quality of fish habitat; decrease in carp and increase in smallmouth bass abundance in former impoundment	Nelson and Pajak 1990, Kanehl et al. 1997

Note: n.d., not defined.

- a. The life span of the dam reflects the total time a dam has been present.
- b. Dam sizes are estimates because many studies did not explicitly state whether height reflected either hydraulic or structural measurements. Impoundment size is based on surface area. Length and impoundment size were not defined in many studies.
- c. For most studies the physical changes are descriptive rather than quantitative.
- d. Partial removal.
- e. Staged removal to be completed in 2003.

on the total volume and particle size distribution of sediment within the impoundment.

The effects of dam removal on biogeochemical processes have varied among studies, probably because of variations in key physical characteristics of different systems. For instance, Stanley and Doyle (2002) studied the impoundment upstream from Rockdale Dam on Koshkonong Creek (WI), which was dominated by fine sediment. Prior to removal, the impoundment retained some forms of phosphorus (P) and was a sink for nitrate; after removal, there was a net export of P-rich sediments to downstream reaches (Stanley and Doyle 2001). Stanley and Doyle (2002) predict that nitrate concentrations will decrease in the former impoundment because of greater sediment-water contact resulting from channel widening, but many months could elapse before measurable declines are evident. In contrast, Bushaw-Newton et al. (2001) studied a small impoundment with little fine-sediment accumulation and a very short hydraulic residence time (approximately 2 hours; calculated as impoundment volume/discharge) on the Manatawny Creek. They observed no significant upstream-downstream differences in dissolved oxygen, temperature, or most forms of nitrogen (N) and P, either before or after dam removal. They proposed that the likelihood of observing impoundment-mediated transformations of these N and P cycles was ultimately related to the depth and hydraulic residence time of the impoundment, which influence not only the magnitude of fine-sediment accumulation but also the potential for thermal stratification and the development of an anoxic hypolimnion.

Biotic responses to dam removal have often been large and rapid. Some of the most dramatic changes stem from the removal of the dam as an obstruction to upstream movement by migratory fish. Within a year after the removal of Edwards Dam on the Kennebec River, large numbers of American eel (*Anguilla rostrata*), alewife (*Alosa pseudoharengus*), Atlantic and shortnose sturgeon (*Acipenser oxyrinchus* and *A. brevirostrum*), and striped bass (*Morone saxatilis*) were observed in upstream habitats that had been inaccessible to these species for more than 150 years (O'Donnell et al. 2001). Two years after removal, more than 1000 larval and juvenile American shad (*Alosa sapidissima*) were collected in the newly accessible reach, and many of these appear to be derived from wild stocks that have migrated upstream (M. O'Donnell, Maine Department of Marine Resources, Augusta, ME, personal communication, 2001). Similar responses by migratory species have been observed following the removal of dams on Bear Creek, Oregon (Smith et al. 2000); Mad River, California (Winter 1990), Neuse River, North Carolina (Bowman 2001); and Clearwater River, Idaho (Shuman 1995). Of course, migratory species are not always present downstream from a dam that is being removed, especially when dams located farther downstream obstruct their upstream movements (see, e.g., Horwitz et al. 2001).

Even in the absence of migratory fish, dam removal permits resident fish species to extend their movements through

out the system. This pattern was observed in the Chipola River, Florida (Estes et al. 1993, Hill et al. 1994); Pine River system, Michigan (Burroughs et al. 2001); Milwaukee River, Wisconsin (Nelson and Pajak 1990, Kanehl et al. 1997); and the Baraboo River system, Wisconsin (Catalano et al. 2001) (table 1). For example, within days or weeks after breaching of the Manatawny Creek Dam, fish that had been tagged downstream from the dam prior to removal were collected in the former impoundment and subsequently observed 1 km upstream (Horwitz et al. 2001). Many studies have also described a general shift from lentic (still water) to lotic (flowing water) species in the former impoundment, such as from carp (*Cyprinus carpio*) to smallmouth bass (*Micropterus dolomieu*) in the Milwaukee River (Nelson and Pajak 1990, Kanehl et al. 1997). Other potential responses to the reversal of dam impacts, including changes in predation on downstream migrants (Zimmerman and Ward 1999) and changes in genetic and population structure (Jager et al. 2001, Neraas and Spruell 2001), have not yet been observed in actual dam removal studies.

Other organisms whose movements are less likely to be hindered by dams can also show dramatic responses to dam removal. For instance, species of benthic algae and macroinvertebrates that were rare or absent within the impoundment in Manatawny Creek increased in abundance within months after dam removal, transforming this zone from a lentic to lotic environment (Hart et al. 2001). Similar results for algae have been observed in Kennebec River, Maine (Casper et al. 2001), and for macroinvertebrates in Baraboo River, Wisconsin (Stanley et al. 2002), and in Turtle Creek, Wisconsin (Pollard and Reed-Anderson 2001).

Given the small number of dam removal studies, as well as the wide range of observed outcomes, we cannot yet draw general conclusions about the range, magnitude, and trajectory of expected ecological responses. Several other factors limit our ability to draw more robust conclusions:

- Most studies are of only a few components of the system (e.g., fish or sediment), rather than an integrated assessment of ecological responses.
- Some studies have relied on qualitative observations rather than quantitative measurements of responses.
- The sampling designs used to make inferences about dam removal effects are often limited by inadequate spatial and temporal replication.
- Dam removal usually causes many abiotic factors to change simultaneously (e.g., flow, sediment transport, water temperature), thereby hampering the identification of causal pathways that govern observed responses.

Improved understanding will require not only that these limitations be overcome but also that a greater focus be placed on how responses to removal vary with dam type, river characteristics, and watershed setting.

Predictions based on ecological effects of existing dams. A simpler, alternative procedure for predicting dam removal responses is to assume that the ecological impacts of an existing dam can be reversed once the dam is removed; we examine the validity of this assumption below. This method seeks to identify the expected ecological conditions, or restoration endpoints, that would exist after a sufficient time period has elapsed following dam removal to permit complete recovery. The approach is more limited than analyses of actual dam removals, however, because it cannot predict the time course of ecological responses. Some useful insights regarding the sequence and rate of these responses can potentially be gained from Petts (1984), who examined various time scales at which different physical, chemical, and biological characteristics responded to the construction of dams.

A central challenge in applying this approach is to determine the type and magnitude of impacts caused by an existing dam. For example, dams vary greatly in size, operation, and watershed setting, and this potentially creates large differences in their ecological impacts (Poff and Hart 2002). Unfortunately, ecologists have not yet studied a wide enough range of dam types to make accurate predictions about the effects of such variation on the structure and function of river ecosystems. Most studies have focused on the ecological effects of large storage dams, which clearly have major impacts on rivers (Ward and Stanford 1979, Petts 1984, Collier et al. 1996). Yet most dams being removed are small, and the effects of small dams may be quite different from those of large dams (Benstead et al. 1999, Poff and Hart 2002). Despite the fact that small, human-made dams have received little study, some insights about their ecological effects can be gained from research on natural analogs of small dams (box 1). For example, beaver (*Castor canadensis*) dams often cause large changes in aquatic habitat types and biogeochemical cycles as well as moderate changes in sediment transport, but they usually have smaller effects on downstream flow regimes. In contrast, waterfalls probably have negligible effects on most ecosystem characteristics, but they can be potent barriers to the upstream movements of fish.

The ecological effects of small human-made dams are likely to be intermediate between the effects of various small natural dams and those of large human-made dams. Figure 2 explores how various ecosystem attributes (e.g., flow regime, sediment transport, biotic migration) may be affected by different types of human-made dams and natural analogs of small dams. For example, beaver dams and small mill dams probably have qualitatively similar effects on nutrient cycling, habitat, and biotic migration, but the range and magnitude of beaver dam effects are presumably reduced because of their porosity and intermittent breakage. Similarly, both small mill dams and large flood control dams can potentially affect flow and temperature regimes, but the impact of the latter structures generally is magnified because of their greater storage capacity, hydraulic residence time, and tendency to stratify thermally. Indeed, recent studies support the idea that small, human-made dams have reduced effects on

thermal regimes (Newcomb 1998, Lessard 2000) and flow regimes (Magilligan and Nislow 2001), compared with large storage dams.

Ultimately, the ability to predict ecological responses to dam removal from a knowledge of existing dam effects could be greatly improved by studying a broader range of dam sizes and types, especially smaller dams. For example, simple scaling considerations may facilitate the prediction of dam effects on some ecosystem attributes as a function of dam and river characteristics, such as the effect of dam height on fish blockage. For this approach to yield useful predictions, however, we need to determine whether the ecological effects of existing dams are actually reversible.

Are the impacts of dams reversible? Given a sufficient amount of time, many of the ecological impacts that dams have on rivers are likely to be largely reversed following dam removal. To date, however, no studies of dam removal have continued long enough to determine the response rates of all ecosystem components. The time course and sequence of recovery will also differ among rivers, dam types, and climatic settings, which must be accounted for to develop realistic expectations about restoration outcomes. Moreover, future

		ECOSYSTEM ATTRIBUTE					
		Flow Regime	Temperature Regime	Sediment Transport	Biogeochemistry	Biotic Migration	Habitat
NATURAL ANALOGS	Waterfall	■	■	■	■	◐	■
	Debris Dam	○	○	○	○	○	○
	Beaver Dam	◐	◐	◐	◐	◐	◐
HUMAN-MADE	<0.5 m height	○	○	○	○	○	○
	1-5 m height (mill dams, weirs, diversion dams)	◐	◐	◐	◐	◐	◐
	>15 m height (water supply, hydro-power, flood control)	●	●	●	●	●	●
EFFECTS		■ NONE	○ SMALL	◐ MODERATE	● LARGE		

Figure 2. Hypothetical relationship between dam type and various ecosystem attributes. Dam types include both natural dams (generally small) and human-made dams of varying heights and operations. The effects level can be defined in terms of the magnitude of change (e.g., the difference between the maximum annual downstream temperature in the presence vs. absence of the dam), the spatial extent of change (e.g., the length of the downstream zone in which temperatures are altered), and the temporal duration of change (e.g., the time interval between beaver dam failures during which biotic migration is obstructed). Thus, the effect of a 1 m mill dam on downstream temperatures is reduced compared with a 50 m flood control dam with a hypolimnetic release, in terms of both the absolute temperature change and the downstream distance at which such changes are manifested.

Box 1. Natural analogs to small dams: Similarities and differences

Natural analog	Small dam comparison
Debris dams	Debris dams alter stream flow, habitat structure, and particulate transport (Bilby and Likens 1980, Wallace and Benke 1984). Debris dams are typically small (< 1 m), porous, and relatively ephemeral.
Beaver dams	Beaver dams have the potential to alter the hydrology, channel geomorphology, biogeochemistry, and productivity of a stream ecosystem (Naiman et al. 1988). Beaver dams may be short-lived, but some dams may exist for decades, and beaver populations may maintain dams at various sites within a watershed over long periods. There is a general shift in the biota of these impoundments from lotic to lentic (Naiman et al. 1988, Snodgrass and Meffe 1998), and fish passage may be blocked (Avery 1992). Beavers alter the riparian areas by cutting mature trees for both dam building and food, which, in turn, opens the surrounding canopy, alters the litter input to the stream, and in many cases causes a shift in vegetation from tree to shrub (Naiman et al. 1988). Most beaver dams are small (< 2 m), semiporous, and subject to intermittent periods of flow between partial breaks and repair.
Landslides	Although there has been increasing attention to landslides as geomorphic agents (Naiman et al. 2000), there has been less attention to impoundment and downstream effects.
Waterfalls	Waterfalls can block fish passage, in some cases providing upstream refuges from introduced species. Impoundment and downstream effects depend on the precise geological conditions of the falls.
Lake outlets	Reservoirs create many of the major impacts on downstream reaches. Lake outlets provide a natural analog to many of these effects, without effects of blockage. Sedimentation, reduced downstream transport of coarse sediments, increased residence time (and consequent geochemical effects), increased primary production and downstream export of plankton, stratification, and support of lentic species occur in both lakes and reservoirs. Increased abundance of filter-feeding macroinvertebrates (e.g., hydropsychid caddis flies, black fly larvae) has been demonstrated in both lake outlet and tailwater locations (Richardson and Mackay 1991). Reservoirs are often very different from natural lakes (due to hypolimnetic releases, significant flow regulation and manipulation of reservoir levels, dendritic topography, etc.), but outlets may provide analogs for smaller, run-of-river dams. More information is needed, however, on the ecological, geochemical, and geomorphic effects of lakes on outlet streams.

research may identify management practices (e.g., timing of dam breaching, sediment management, control of exotic species, riparian planting, improving in-stream habitat, or reintroducing desirable organisms) that can increase recovery rates in some circumstances.

Our previous discussion of observed responses to dam removal has direct relevance to the question of ecological reversibility, and it is useful to review some of the major factors likely to influence the recovery process and the potential for reversibility. For example, soon after dam removal, many features of the river's flow regime may be restored. The effects of a dam on water quality and thermal regime often are rapidly reversed because of decreased hydraulic residence time and stratification, which in turn affect sedimentation and nutrient cycling. The time course of geomorphic adjustments to dam removal varies with the sediment types within the former impoundment and the ability of the river to transport that sediment (Pizzuto 2002, Stanley and Doyle 2002). Several years to more than a decade may be needed to reestablish an equilibrium channel. The slower time scale of geomorphic change may also control the rate of change in other

ecosystem attributes. For instance, sediment-bound nutrients in the former impoundment may continue to affect water quality. Several important ecosystem features (e.g., pattern of floodplain inundation and habitat characteristics) are strongly affected by hydrology, which in turn depends on channel morphology, so the restoration of these ecological attributes follows the time scale of channel changes.

Biota respond to the physical removal of the barrier, as well as to changes in water chemistry, habitat, and flow regime. The potential for recovery of various taxa following dam removal varies markedly, depending in part on their ability to colonize and thrive in new habitats. For instance, algae, some higher plants, and many invertebrates may quickly colonize the former impoundment and downstream reaches by means of downstream transport. Plant seeds may also be present in impoundment sediments (Shafroth et al. 2002). Although initial colonization may be rapid, population recovery in the former impoundment and downstream reaches ultimately depends on restoration of habitat conditions (e.g., temperature, substrate, topography, large woody debris) that are strongly influenced by channel morphology, flow regimes, and

riparian vegetation. The time course of recovery is influenced by individual and population growth rates (e.g., benthic algae recover more quickly than riparian trees). Similarly, unionid mussels may colonize slowly because of their relatively slow growth rates and specific habitat requirements, as well as their dependence on fish for dispersal (Watters 1996).

Studies of biotic recovery have focused particular attention on the elimination of blockage to anadromous fish migrations. These species are quite mobile and can move many miles upstream from the dam site within weeks to months following removal. Recolonization of migratory species may occur slowly or not at all without active introduction programs, however, if migration depends on the existence of stocks that have imprinted on natal streams or that require cues based on conspecific pheromones (Vrieze and Sorensen 2001).

Determining whether dam impacts are reversible not only requires a focus on the processes that contribute to ecological recovery following dam removal, it also depends on how the concept of reversibility is defined. To some, reversal may denote the attainment of ecological conditions that existed before the dam was constructed or that are present in regional reference sites (NRC 1992). Given the widespread occurrence of beaver dams in North America prior to European settlement (Naiman et al. 1988), however, some qualitative effects of dams undoubtedly existed long before humans constructed dams (see box 1 and figure 2). More important, dams are usually not the only factor impairing river ecosystems, which can lead to unrealistic expectations about recovery following dam removal. Many dams are located in watersheds that are stressed by other forms of habitat alteration (e.g., channelization, loss of riparian vegetation) as well as a diverse array of point source and non-point source pollutants.

Mechanistic and empirical models for predicting responses to dam removal. Given the wide range of possible impacts of dams and dam removal, and the complex ways these impacts depend on dam, river, and watershed characteristics, models can potentially serve as an important predictive tool. For example, conceptual models of sediment transport provide a valuable framework for understanding changes in channel form following dam removal, although precise quantitative models do not yet exist (Pizzuto 2002). Similarly, population fragmentation models used to predict dam impacts on migratory and resident fishes (Jager et al. 2001) may help in evaluating population consequences of dam removal.

Simple models are needed that can predict the occurrence and magnitude of important impoundment processes (e.g., sedimentation, stratification, and nutrient transformations) on the basis of characteristics such as dam and reservoir dimensions or hydraulic residence time (volume/discharge). For example, various formulations of lake nutrient models relate concentrations to geometric and hydraulic parameters such as the surface overflow rate, calculated as either discharge/surface area or depth/hydraulic residence time (Chapra and Reckhow 1983, Reckhow and Chapra 1983). The occurrence

of thermal stratification is related to depth, wind speed, water velocity, and heat flux (Condie and Webster 2001). For instance, surface area/depth has been used as a simple predictor of susceptibility to stratification in lakes (Stefan et al. 1996). Although these indices may correctly rank some relative effects of large and small dams, no single parameter of dam size can properly scale all dam effects. For example, models designed to incorporate the effects of river inflows and outflows and the complex topography of impoundments usually require more complex terms for advection (e.g., river-run models; Chapra and Reckhow 1983) and spatial subdivision (Schnoor 1996).

Dam removal and river restoration

Risk assessment framework for evaluating the potential effects of dam removal. If dam removal is to become an effective method of river restoration, we must be able to predict the potential benefits of any proposed removal. As discussed above, prior dam removal studies, as well as assessments of existing dam impacts, indicate that the ecological effects of dam removal are likely to vary from project to project because of differences in dam, river, and watershed characteristics. How can we improve the scientific basis for dam removal decisions if ecological responses to removal are so variable?

We propose an ecological risk assessment framework that can be used to account for many of the factors that influence variation in potential responses to dam removal, thereby enhancing our ability to predict those responses. In ecological risk assessment, ecological effects are characterized by determining the potential effects imposed by a stressor, linking these effects to assessment endpoints, and evaluating how effects change with different stressor levels (USEPA 1998). As used above, "effects" are the observed changes in various ecological attributes, and "endpoints" are the broader environmental values or management goals that give context and meaning to the observed effects. This basic framework can be used to evaluate the effects of dam removal by considering dams as stressors and dam size (or another measure that accounts for dam, river, and watershed characteristics) as a measure of stressor level. The ecological effects of dam removal can then be determined as a function of variation in dam and watershed characteristics. Application of this framework allows an assessment of the potential benefits of dam removal across a range of dam and river or watershed conditions in the context of specific watershed management goals (e.g., fisheries production, water quality enhancement, habitat improvement). In turn, this information can be used to help select and prioritize dam removal projects, thereby maximizing the effectiveness of dam removal in river restoration.

Central to this approach is the determination of how the ecological effects of dam removal vary across a range of dam and watershed characteristics. In the language of ecological risk assessment, the relationship between a river's ecological integrity (response) and a particular dam or watershed char-

acteristic (stressor) such as dam size is called a stressor–response relationship (figure 3). When a reference condition is considered together with a stressor–response relationship, the maximum “potential” benefit of a particular dam removal can be determined. For example, the maximum potential benefit for curve 2 at a dam stressor level of x is shown by the arrow (figure 3). Given the shape of the stressor–response relationship and the magnitude of the stressor, a maximum potential benefit can thus be estimated for any ecological attribute resulting from the removal of a dam. Achieving the maximum potential benefit assumes complete recovery of the system to predam conditions, which may not always be possible. As in all restoration, selection of reference conditions is extremely important, and the methods used to determine reference conditions are likely to differ among ecological attributes. For example, potential reference conditions could be based on upstream conditions, historical

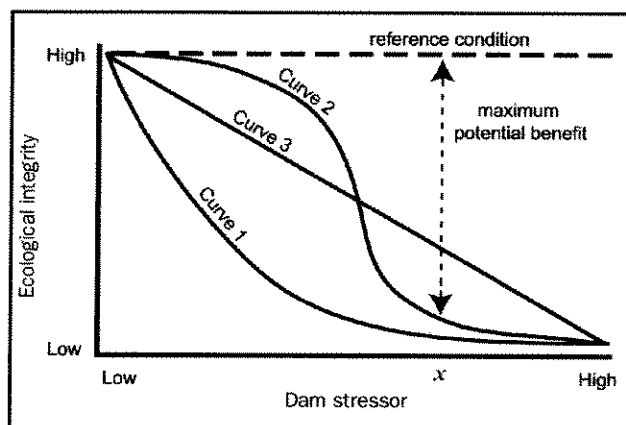


Figure 3. Generalized linear and nonlinear relationships between dam stress (stressor) and ecological integrity (response). Dam stress may be characterized as crest height or dam width, or it may be scaled according to various river and watershed characteristics. Ecological integrity can refer to any physical, chemical, or biological attribute of the river system. Many nonlinear forms of this relationship are possible. For a given stressor level, the maximum potential benefit of dam removal is shown as the difference between the stressor–response curve and a reference condition.

conditions prior to dam construction, or conditions at regional reference sites.

To apply this risk assessment framework, it is important to understand how the effects of dam removal differ across a range of dam and watershed characteristics, and to recognize that the shape of the stressor–response relationship varies with different ecological effects and endpoints. For example, three potential relationships are shown in figure 3, although many more are possible. Curve 1 depicts a nonlinear relationship where the reduction in ecological integrity with a unit increase in dam stress is greatest at low dam stress levels. This could potentially describe the effect of dam height on the upstream

migration of river herring, whose passage is obstructed by even the smallest of dams and culverts. Curve 2 also shows a nonlinear relationship, but in this case there are two thresholds rather than one. This may be representative of changes in temperature or various biogeochemical processes affected by thermal stratification. For instance, a lower threshold in depth or hydraulic residence time has to be exceeded before stratification begins, and once the upper threshold for this relationship is exceeded, no further changes in stratification occur. Lastly, curve 3 shows a simple linear relationship where the ecosystem response is directly proportional to the dam stress. It is not yet clear what components of ecological integrity might be linearly related to particular dam stressors. Note that if the stressor–response relationship is nonlinear, then the potential benefits of dam removal vary in a complex way depending on dam and watershed characteristics. For example, if the stressor–response relationship is similar to curve 2, then the removal of dams with stressor levels below the lower threshold may yield relatively small ecological benefits.

Currently, the shape of these stressor–response relationships is not well known, but relationships can be developed using any of the three approaches for predicting ecological responses to dam removal discussed previously. Establishing relationships based on observation of completed dam removal projects, however, would require comprehensive studies lasting many years at numerous sites across a gradient of dam, river, and watershed characteristics. This may not be possible for a number of years, because so few studies of dam removal have been completed. Likewise, the development of mechanistic models describing ecosystem structure and function has not yet advanced to the stage where they can be readily used to predict stressor–response relationships. Thus, we suggest that the best opportunity at the present time for developing stressor–response relationships and predicting restoration outcomes is to examine the effects of existing dams. We are currently quantifying these relationships across a range of dam and river types in the Mid-Atlantic region, with the ultimate goal of using this approach to prioritize dam removals so that restoration benefits are maximized. We also strongly encourage studies of actual dam removals and the development of better mechanistic models to help define stressor–response relationships.

The successful application of this risk assessment approach depends on the ability to extrapolate from the known ecological effects of a sample of dams to predict the effects of other dams being considered for removal. This requires that ecological responses to the removal of a particular dam are similar to the responses that would occur for other dams of similar size, operational type, hydraulic residence time, drainage area, and so on. Given the potential for marked geographic variation in dam impacts and river responses, this requirement is more likely to be met in a restricted physiographic region. We also need to identify appropriate measures or scaling factors that can quantify the relative stress imposed by a given dam on a particular river. Although dam height is clearly important, the impact of a dam on a river is also likely to vary

depending on river characteristics such as flow regime, channel form, sediment transport, and nutrient status. A number of different measures may also prove useful in predicting dam impacts, including the impoundment's hydraulic residence time, ratio of dam height to a reference channel width, degree of flow modification, and frequency of thermal stratification within impoundments.

Potential adverse effects of dam removal. The risk assessment framework can help guide dam removal decisions based on expected restoration outcomes, but we must also be mindful that dam removal can have negative effects. For example, ecological impacts sometimes result from large movements of sediment (especially when contaminants are present). An ongoing dam removal study on Kettle River, Minnesota, revealed declines in downstream mussel populations following a dam removal; the declines were attributed to the export of coarse sediment from the former impoundment. The extent to which these effects were offset by restored fish host access to upstream areas is unclear (L. Aadlund, Minnesota Department of Natural Resources, Fergus Falls, MN, personal communication, 2001) (table 1). In the Baraboo River system (WI), the removal of several dams improved fish habitat quality within the former impoundments but decreased fish habitat downstream (Catalano et al. 2001). Substantial reductions in the abundance of several nonmigratory fish species were observed immediately downstream from the former dam following several major sediment transport events that occurred after the removal of Manatawny Creek Dam (Horwitz et al. 2001). These negative effects were probably due to habitat modification (e.g., sediment accumulation in pools and parts of riffles, and sediment scouring in other parts of riffles) that caused fish to move to other areas. Partial recovery of fish assemblages in these riffles was observed a year after removal, and full recovery is likely once sediment from the former impoundment has moved downstream.

Other adverse effects may include reductions in wetland habitat or groundwater recharge, as well as shifts in species abundance and distribution. For example, declines in recreationally important biota have been observed for several removals. In addition, despite the recommended usage of dam removal to eliminate barriers to fish movement, there are some situations where removal could potentially increase the chances that exotic species presently blocked by dams could invade upstream habitats. For instance, dam removal could permit sea lamprey (*Petromyzon marinus*) to invade various rivers that drain into the Great Lakes (Dodd 1999), or flathead catfish (*Pylodictis olivaris*) could move upstream in various rivers of the Atlantic coastal plain (T. Kwak, North Carolina State University, Raleigh, NC, personal communication, 2001).

Some of these adverse responses to dam removal are probably transient, however, and might be considered analogous to the short-term impairment of human performance that often occurs during the recuperative period following surgery.

Other impacts (e.g., those due to sediment transport) are perhaps best evaluated in the context of natural disturbance regimes. For example, the magnitude, timing, and duration of sediment effects associated with dam removal may be no different from those caused by natural variations in sediment transport. Alternatively, suspended and bedload transport following dam removal may greatly exceed natural levels, thus producing ecological changes far beyond those caused by natural disturbance. Some undesirable effects of dam removal can potentially be reduced by developing improved restoration practices, particularly with respect to sediment management (ASCE 1997). For instance, inexpensive but effective methods are needed to assess and mitigate contaminant risks. These assessments should include a review of the historical usage of the watershed, as well as an analysis of the type and grain size of sediments in the impoundment (Bushaw-Newton et al. 2001). Even when contaminants are absent, we need to know how much sediment can safely be released, and during what seasons, to minimize downstream impacts. Such information could guide efforts to control sediment releases by removing dams incrementally (ASCE 1997), or by planting riparian vegetation to stabilize sediments (e.g., Shafroth et al. 2002). In some cases, species of special concern may be particularly vulnerable during dam removal. For example, some species of fish or mussels may be stranded as the impoundment is drawn down, which may create a need for inexpensive methods of collecting and relocating these species.

Comprehensive watershed management and dam removal. Dam removal may be the most direct and effective method for eliminating the negative effects of dams on the structure and function of river ecosystems, but it is only one of several dam management alternatives. Depending on the particular dam, these options may include no action, structural repair, dam removal, or changes to dam operations (ASCE 1997). The last option potentially involves a variety of actions, such as the installation or improvement of devices to allow fish passage, modification of water release practices to create more natural flow and sediment transport regimes (Webb et al. 1999), or the enhancement of downstream water quality by aeration and temperature modification (Higgins and Brock 1999). Wherever possible, objective criteria and formal models should be used to evaluate the costs and benefits of these dam management alternatives (Whitelaw and MacMullan 2002). In cases in which dam removal is not considered a viable option (e.g., for economic or political reasons), various reoperation strategies have the potential to reduce some (but not all) of the negative effects that dams can have on ecological integrity.

In most watersheds, however, successful river restoration will require a focus on more than just the problems created by dams. Effective watershed management depends on an integrative approach that identifies the full range and types of stressors impairing the ecosystem and implements controls and practices to reduce these impacts. Because many streams

and rivers are impaired by more than one kind of stressor, a coordinated effort is clearly needed. For example, a particular river system may potentially be impaired by acid mine drainage or sediment from logging operations in its headwaters, by hydropower dams and nutrient-enriched runoff from agricultural fields in its middle reaches, and by high contaminant levels emanating from urban sources (e.g., wastewater effluent as well as stormwater runoff) near its mouth. Dam removal may prove to be a particularly useful method for reducing some forms of ecosystem impairment, but it needs to be considered as part of a broad, watershed-scale management plan (Stanford et al. 1996). To accomplish effective river restoration, dam removal will likely need to be coupled with other protection and restoration practices.

Conclusion

Over the last few years, there has been an increasing focus on the potential value of dam removal in river restoration by ecological researchers, watershed managers, and policymakers. The growing number of scientific studies provides an important opportunity to learn how better to manage watersheds and improve our understanding of the science of river restoration. Increases in the number of completed and prospective dam removals also create a significant challenge, however. Without an integrated scientific framework within which to predict and examine potential ecological responses, there is the danger that these projects will proceed without sufficient learning to improve the effectiveness of future removals. By placing some of our current knowledge in a risk assessment framework, scientists, managers, and other stakeholders can begin to understand and predict how dam removal can be used most effectively to achieve watershed restoration goals.

Acknowledgments

We thank Scott Carney and Andrew Newton as well as many of our Patrick Center colleagues for useful discussions about these issues. Financial support from a Pennsylvania Department of Environmental Protection Growing Greener Grant, a private foundation, and the Patrick Center's Research Endowment Fund allowed us to examine ecosystem responses to the removal of the Manatawny Creek Dam and to develop many of the ideas in this article. The views expressed herein are those of the authors and do not necessarily reflect the views of the Pennsylvania Department of Environmental Protection.

Becky Brown, Tom Coon, Matt Greenstone, and LeRoy Poff offered valuable feedback on earlier drafts of this manuscript. Robin Davis offered outstanding editorial assistance. The Pew Charitable Trusts provided support that facilitated the completion of this paper.

References cited

- [AR/FE/TU] American Rivers, Friends of the Earth, Trout Unlimited. 1999. Dam Removal Success Stories: Restoring Rivers through Selective Removal of Dams That Don't Make Sense. Washington (DC): AR/FE/TU.
- [ASCE] American Society of Civil Engineers, Task Committee on Guidelines for Retirement of Dams and Hydroelectric Facilities of the Hydropower Committee of the Energy Division. 1997. Guidelines for the Retirement of Dams and Hydroelectric Facilities. New York: ASCE.
- Avery EL. 1992. Effects of removing beaver dams upon a northern Wisconsin brook trout stream. Madison (WI): Wisconsin Department of Natural Resources. Study no. 406.
- Babbitt B. 2002. What goes up, may come down. *BioScience* 52: 656–658.
- Bednarek AT. 2001. Undamming rivers: A review of the ecological impacts of dam removal. *Environmental Management* 27: 803–814.
- Benstead JP, March JG, Pringle CM, Scatena FN. 1999. Effects of a low-head dam and water abstraction on migratory tropical stream biota. *Ecological Applications* 9: 656–668.
- Bilby RE, Likens GE. 1980. Importance of organic debris dams in the structure and function of stream ecosystems. *Ecology* 61: 1107–1113.
- Born SM, Genskow KD, Filbert TL, Hernandez-Mora N, Keefer ML, White KA. 1998. Socioeconomic and institutional dimensions of dam removals: The Wisconsin experience. *Environmental Management* 22: 359–370.
- Bowman SW. 2001. American shad and striped bass spawning migration and habitat selection in the Neuse River, North Carolina. Masters thesis. North Carolina State University, Raleigh, NC.
- Burroughs BA, Hayes DB, Mistak JL. 2001. Dam removal effects on fisheries resource and habitat in a Michigan coldwater stream. *Bulletin of the North American Benthological Society* 18: 181.
- Bushaw-Newton KL, Ashley JT, Boettner AR, DeAlteris J, Kiry P, Kreeger DA, Raksany D, Velinsky DJ. 2001. The Manatawny Creek Dam removal: Biogeochemical processes and sediment contaminants. *Bulletin of the North American Benthological Society* 18: 172.
- Casper AF, Thorp JH, Davies SP, Courtemanch DL. 2001. Initial response of benthic primary consumers to dam removal on the Kennebec River, Maine. *Bulletin of the North American Benthological Society* 18: 177.
- Catalano MJ, Zozek MA, Pellett TD. 2001. Fish-habitat relations and initial response of the Baraboo River fish community to dam removal. *Bulletin of the North American Benthological Society* 18: 177.
- Chapra SC, Reckhow KH. 1983. *Engineering Approaches for Lake Management, Vol. 2: Mechanistic Modeling*. Ann Arbor (MI): Ann Arbor Science.
- Collier M, Webb RH, Schmidt JC. 1996. *Dams and Rivers: A Primer on the Downstream Effects of Dams*. Menlo Park (CA): US Geological Survey. Circular no. 1126.
- Condie SA, Webster LT. 2001. Estimating stratification in shallow water bodies from mean meteorological conditions. *Journal of Hydraulic Engineering* 127: 286–292.
- Dodd HR. 1999. The effects of low-head lamprey barrier dams on stream habitat and fish communities in tributaries of the Great Lakes. Master's thesis. Michigan State University, East Lansing, MI.
- Estes JR, Myers RA, Mantini L. 1993. Fisheries Investigations of Newnan's Lake. Tallahassee (FL): Florida Game and Fresh Water Fish Commission. Wallop-Breaux Project F-55-6, Study IV.
- Hart DD, Finelli CM. 1999. Physical-biological coupling in streams: The pervasive effects of flow on benthic organisms. *Annual Review of Ecology and Systematics* 30: 363–395.
- Hart DD, et al. 2001. The Manatawny Creek Dam removal: Species and community characteristics. *Bulletin of the North American Benthological Society* 18: 172–173.
- Higgins JM, Brock WG. 1999. Overview of reservoir release improvements at 20 TVA dams. *Journal of Energy Engineering* 125: 1–17.
- Hill MJ, Long EA, Hardin S. 1994. Effects of dam removal on Dead Lake, Chipola River, Florida. *Proceedings of the Annual Conference of South-eastern Association Fish and Wildlife Agencies* 48: 512–523.
- Horwitz RJ, Overbeck P, Perillo J, Bushaw-Newton K. 2001. Effects on fish populations of removal of a dam on Manatawny Creek (Schuylkill River drainage, Pottstown, Pennsylvania). Paper presented at the Annual Meeting of the American Fisheries Society; 19–23 August 2001; Phoenix, AZ.
- Jager HI, Chandler JA, Lepla KB, Van Winkle W. 2001. A theoretical study of river fragmentation by dams and its effects on white sturgeon populations. *Environmental Biology of Fishes* 60: 347–361.

- Johnson SL. 2001. Kettle River Dam removal: Impacts of sediment on downstream mussel populations. Paper presented at a meeting of the Freshwater Mollusk Conservation Society; March 2001; Pittsburgh, PA.
- Johnson TE, Pizzuto J, Egan J, Bushaw-Newton K, Hart D, Lawrence J, Lynch E. 2001. The Manatawny Creek Dam removal: Project overview and geomorphic characteristics. *Bulletin of the North American Benthological Society* 18: 121–122.
- Kanehl PD, Lyons J, Nelson JE. 1997. Changes in the habitat and fish community of the Milwaukee River, Wisconsin, following removal of the Woolen Mills Dam. *North American Journal of Fisheries Management* 17: 387–400.
- Lessard JL. 2000. Temperature effects of dams on coldwater fish and macroinvertebrate communities in Michigan. Master's thesis. Michigan State University, East Lansing, MI.
- Magilligan FJ, Nislow K. 2001. Long-term changes in regional hydrologic regime following impoundment in a humid-climate watershed. *Journal of the American Water Resources Association* 37: 1551–1570.
- Naiman RJ, Johnston CA, Kelley JC. 1988. Alteration of North American streams by beaver. *BioScience* 38: 753–762.
- Naiman RJ, Bilby RE, Bisson PA. 2000. Riparian ecology and management in the Pacific coastal rain forest. *BioScience* 50: 996–1011.
- [NRC] National Research Council. 1992. *Restoration of Aquatic Ecosystems*. Washington (DC): National Academy Press.
- Nelson JE, Pajak P. 1990. Fish habitat restoration following dam removal on a warm water river. Pages 57–65 in *American Fisheries Society, North Central Division, Rivers and Streams Technical Committee Symposium Proceedings: The Restoration of Midwestern Stream Habitat*; 4–5 December 1990; Minneapolis, MN.
- Neraas LP, Spruell P. 2001. Fragmentation of riverine systems: The genetic effects of dams on bull trout (*Salvelinus confluentus*) in the Clark Fork River system. *Molecular Ecology* 10: 1153–1164.
- Newcomb TJ. 1998. Productive capacity of the Betsie River watershed for steelhead smolts. PhD dissertation. Michigan State University, East Lansing.
- O'Donnell M, Gray N, Wipfelhauser G, Christman P. 2001. Kennebec River diadromous fish restoration annual progress report—2000. Augusta (ME): Maine Department of Natural Resources.
- Pawloski JT, Cook LA. 1993. Sallings Dam drawdown and removal. Paper presented at the Midwest Region Technical Seminar on Removal of Dams, Association of State Dam Safety Officials; 30 September–1 October 1993; Kansas City, MO.
- Pejchar L, Warner K. 2001. A river might run through it again: Criteria for consideration of dam removal and interim lessons from California. *Environmental Management* 28: 561–575.
- Petts GE. 1984. *Impounded Rivers: Perspectives for Ecological Management*. New York: John Wiley and Sons.
- Pizzuto JE. 2002. Effects of dam removal on river form and process. *BioScience* 52: 683–691.
- Poff NL, Hart DD. 2002. How dams vary and why it matters for the emerging science of dam removal. *BioScience* 52: 659–668.
- Poff NL, Allan JD, Bain MB, Karr JR, Prestegard KL, Richter BD, Sparks RE, Stromberg JC. 1997. The natural flow regime: A paradigm for conservation and restoration of river ecosystems. *BioScience* 47: 769–784.
- Pollard AI, Reed-Anderson T. 2001. Benthic invertebrate community change following dam removal in a small Wisconsin stream. *Bulletin of the North American Benthological Society* 18: 173.
- Power ME, Sun A, Parker G, Dietrich WE, Wootton JT. 1995. Hydraulic food-chain models. *BioScience* 45: 159–167.
- Reckhow KH, Chapra SC. 1983. *Engineering Approaches for Lake Management, Vol. 1: Data Analysis and Empirical Modeling*. Ann Arbor (MI): Ann Arbor Science.
- Richardson JS, Mackay, RJ. 1991. Lake outlets and the distribution of filter-feeders: An assessment of hypotheses. *Oikos* 62: 370–380.
- Schnoor JL. 1996. *Environmental Modeling*. New York: John Wiley and Sons.
- Shafroth PB, Friedman JM, Auble GT, Scott ML, Braatne JH. 2002. Potential responses of riparian vegetation to dam removal. *BioScience* 52: 703–712.
- Shuman JR. 1995. Environmental considerations for assessing dam removal alternatives for river restoration. *Regulated Rivers: Research and Management* 11: 249–261.
- Simons RK, Simons DB. 1991. Sediment problems associated with dam removal—Muskegon River, Michigan. Pages 680–685 in *Hydraulic Engineering: Proceedings of the 1991 National Conference of the American Society of Civil Engineers*. New York: American Society of Civil Engineers.
- Smith LW, Dittmer E, Prevost M, Burt DR. 2000. Breaching of a small irrigation dam in Oregon: A case history. *North American Journal of Fisheries Management* 20: 205–219.
- Snodgrass JW, Meffe GK. 1998. Influence of beavers on stream fish assemblages: Effects of pond age and watershed position. *Ecology* 79: 928–942.
- Stanford JA, Ward JV, Liss WJ, Frissell CA, Williams RN, Lichatowich JA, Coutant CC. 1996. A general protocol for restoration of regulated rivers. *Regulated Rivers: Research and Management* 12: 391–413.
- Stanley EH, Doyle MW. 2001. Phosphorus transport before and after dam removal from a nutrient-rich creek in southern Wisconsin. *Bulletin of the North American Benthological Society* 18: 172.
- . 2002. A geomorphic perspective on nutrient retention following dam removal. *BioScience* 52: 693–701.
- Stanley EH, Luebbe M, Doyle MW, Marshall DW. 2002. Short-term changes in channel form and macroinvertebrate communities following low-head dam removal. *Journal of North American Benthological Society* 21: 172–187.
- Stefan HG, Hondzo M, Fang X, Eaton JG, McCormick JH. 1996. Simulated long-term temperature and dissolved oxygen characteristics of lakes in the north-central United States and associated fish habitat limits. *Limnology and Oceanography* 41: 1124–1135.
- [USEPA] US Environmental Protection Agency. 1998. *Guidelines for Ecological Risk Assessment*. Washington (DC): USEPA. EPA/630/R-95/002F.
- Vrieze LA, Sorensen PW. 2001. Laboratory assessment of the role of a larval pheromone and natural stream odor in spawning stream localization by migratory sea lamprey (*Petromyzon marinus*). *Canadian Journal of Fisheries and Aquatic Sciences* 58: 2374–2385.
- Wallace JB, Benke AC. 1984. Quantification of wood habitat in subtropical coastal plain streams. *Canadian Journal of Fisheries and Aquatic Sciences* 41: 1643–1652.
- Ward JV, Stanford A. 1979. *The Ecology of Regulated Streams*. New York: Plenum Publishing.
- Watters GT. 1996. Small dams as barriers to freshwater mussels (Bivalva, Unionoida) and their hosts. *Biological Conservation* 75: 79–85.
- Webb RH, Schmidt JC, Marzoff GR, Valdez RA. 1999. The 1996 controlled flood in Grand Canyon. Washington (DC): American Geophysical Union. Geophysical Monograph 110.
- Whitelaw E, MacMullan E. 2002. A framework for estimating the costs and benefits of dam removal. *BioScience* 52: 724–730.
- Williams DT. 1977. *Effects of dam removal: An approach to sedimentation*. Davis (CA): US Army Corps of Engineers. Hydrologic Engineering Section. Technical Paper 50.
- Williams JE, Wood CA, Dombek MP, eds. 1997. *Watershed Restoration: Principles and Practices*. Bethesda (MD): American Fisheries Society.
- Winter BD. 1990. *A Brief Overview of Dam Removal Effects in Washington, Oregon, Idaho, and California*. Washington (DC): US Department of Commerce. NOAA Technical Memo NMFS F/NWR-28.
- Zimmerman MP, Ward DL. 1999. Index of predation on juvenile salmonids by northern pike minnow in the Lower Columbia River Basin, 1994–1996. *Transactions of the American Fisheries Society* 128: 995–1007.

PUBLISHER
Richard T. O'Grady
EDITOR-IN-CHIEF
Timothy M. Beardsley
SENIOR EDITOR
Donna Daniels Verdier

PRODUCTION MANAGER / ART DIRECTOR
Herman Marshall

MANUSCRIPT EDITOR
Anthony Nathe

PUBLICATIONS ASSISTANT
Sevon Johnson

Editors: Feature Articles: Cathy Lundmark (features@aibs.org), Washington Watch: AIBS Public Policy Office (publicpolicy@aibs.org), Book Reviews: Paula Mabee, Univ. of South Dakota (pmabee@usd.edu).

Editorial Board: Agriculture: Sonny Ramaswamy; Animal Behavior: Janice Moore; Botany: Lucinda A. McDade; Cell Biology: Randy Wayne; Computers and Biology: Robert V. Blystone; Developmental Biology: Robert Auerbach; Ecology: Scott Collins, Daniel Simberloff; Ecotoxicology: Valery Forbes; Education: Gordon E. Uno; Environmental Policy: Gordon Brown, J. Michael Scott; Genetics and Evolution: Martin Tracey; History and Philosophy: Richard M. Burian; Invertebrate Biology: Kirk Fitzhugh; Landscape Ecology: Monica Turner; Microbiology: Ronald M. Atlas; Molecular Biology: David Hillis; Neurobiology: Cole Gilbert; Population Biology: Ben Pierce; Professional Biologist: Jean Wyld; Statistics: Mark E. Payton; Vertebrate Biology: Harry W. Greene.

Editorial Correspondence: 1444 I Street, NW, Suite 200, Washington, DC 20005; 202/628-1500; Fax: 202/628-1509; e-mail: bioscience@aibs.org. Instructions for preparing a manuscript for *BioScience* can be found at www.aibs.org/bioscience/guide/resources/contributors.html.

Advertising: Kerrie Brooks, Network Publications, 10155 York Rd., Suite 205, Crestridge Corporate Center, Hunt Valley, MD 21030; 410/628-5772; Fax: 410/628-0398; e-mail: bioscience@networkpub.com

BioScience (ISSN 0006-3568) is published monthly by the American Institute of Biological Sciences. Individual membership: Sustaining, \$90/yr; individual, \$70/yr; family, \$90/yr (includes \$36 for *BioScience*); emeritus, \$50/yr; K-12 teacher/administrator, \$45/yr (includes \$22 for *BioScience*); graduate and postdoctoral students, \$40/yr (includes \$21 for *BioScience*); undergraduate and K-12 students, \$20/yr (includes \$15 for *BioScience*); lifetime, \$1,400 (one-time fee). Institutional subscriptions: Domestic, \$254/yr; foreign, \$303/yr. Single copies: \$14 plus shipping and handling for up to 50 copies; volume discounts available for more than 50 (call 703/790-1745, ext. 23). Subscription renewal month is shown in the four-digit year-month code in the upper right corner of the mailing label. © 2002 American Institute of Biological Sciences. All rights reserved. Periodical postage paid at Washington, DC, and additional mailing offices. **POSTMASTER:** Send address changes to *BioScience* Circulation, AIBS, 1313 Dolley Madison Blvd., Suite 402, McLean, VA 22101. Printed in USA. AIBS authorizes photocopying for internal or personal use, provided the appropriate fee is paid directly to the Copyright Clearance Center, 222 Rosewood Drive, Danvers, MA 01923; phone: 978/750-8400; fax: 978/750-4744; Web site: www.copyright.com. To photocopy articles for classroom use, request authorization, subject to conditions thereof, from the Academic Permissions Service at CCC. Each copy must say "© 20__ by the American Institute of Biological Sciences." Statements and opinions expressed in *BioScience* are those of the author(s) and do not necessarily reflect the official positions of the American Institute of Biological Sciences, the editors, the publisher, or the institutions with which the authors are affiliated. The editors, publisher, and AIBS disclaim any responsibility or liability for such material.

BioScience

American Institute of Biological Sciences

Roles for scientists in community-based ecological restoration

You may not have noticed yet, but a community group near you is probably involved in the ecological restoration of a local watershed. Whether their objective is to help remove an abandoned dam, plant streamside vegetation, fence livestock out of streams, or reduce the spread of invasive species, these groups are working hard to improve environmental quality and build a brighter future for their communities. Because many of these restoration projects are initiated by grassroots and volunteer organizations rather than by large government agencies, the participants may not be familiar with the broad range of scientific and technical issues surrounding the nascent field of ecological restoration. Nor are they likely to have major funding to support research. Nonetheless, the thousands of local projects initiated around the country every year offer great opportunities for scientific participation.

Scientists can aid, and potentially benefit from, such community-based restoration in several ways. First, at its core, successful restoration is critically dependent on scientific understanding. Specifically, ecological restoration programs seek to reduce the negative effects of human activities on ecosystems, while enhancing various physical, chemical, and biological processes by which these systems recover from disturbance. Thus, biologists and other scientists can use their knowledge of "how nature works" to help identify the threats to ecological integrity and develop methods for facilitating the recovery of these complex systems.

Scientists can also help ensure that future restoration efforts produce important new understanding of ecological systems. To date, outcomes of many local restoration projects have not even been quantified. To improve this situation, scientists can develop testable hypotheses about the causes of an ecosystem's impairment, as well as the ways in which ecological recovery can be enhanced. Similarly, scientists should look for opportunities to use restoration projects as true experiments and encourage the acquisition of data to determine how ecological systems respond to the restoration "treatment." As more of these objective assessments of restoration outcomes are made, researchers will not only gain new scientific insights but also help determine which restoration practices work best, thereby contributing to more effective ecological restoration.

Scientists who participate in local restoration projects will necessarily communicate with nonscientists about the scientific enterprise and the ways it can be used both to create new knowledge and to help solve real-world problems. The more effectively we can translate the sometimes arcane world of science and explain its relevance to issues of local concern, the more likely that society will value science as a cornerstone of understanding and problem solving. Whether directly or indirectly, such dialogue can also lead to greater public support for scientific research.

Perhaps the best reason for participating in a local restoration project is the personal satisfaction it can offer. Community-based restoration is an inherently optimistic and constructive endeavor, in which citizens are working in their own backyards to help sustain the planet's life support systems. What could be more satisfying than the knowledge that you are adding value to such efforts?

DAVID D. HART

Vice President and Director
Patrick Center for Environmental Research
Academy of Natural Sciences
1900 Benjamin Franklin Parkway
Philadelphia, PA 19103

Effects of Dam Removal on River Form and Process

JIM PIZZUTO

Dams have a profound influence on fluvial processes and morphology. Reservoirs formed by dams drown river channels and trap sediment. Downstream reaches respond to altered flow regimes and reduced sediment supply in varied ways (Williams and Wolman 1984, Collier et al. 1997) that are difficult to predict, although common responses include erosion and lowering of the channel bed (incision) and development of a coarse-grained surface layer (armor) in the riverbed downstream of a dam.

As dam removal continues to gain momentum as a restoration strategy, understanding how a river changes when a dam is removed is becoming increasingly important. Because few detailed geomorphic studies of dam removal have been conducted, however, there is little direct observational basis for predicting the geomorphic effects of dam removal. Furthermore, rivers are complex and fluvial processes often occur over decades or centuries, so predictions are inherently uncertain.

Fortunately for researchers, the processes associated with dam removal also occur naturally. For example, after dam removal the sediment fill in an impoundment is likely to become incised, and an equilibrium channel with a new floodplain is likely to form as sediment evacuated during incision increases the sediment supply to downstream reaches. Natural processes related to incision, floodplain formation, equilibrium channel development, and increased sediment supply have been widely studied by geomorphologists and engineers, providing useful conceptual models for evaluating the geomorphic effects of dam removal (Doyle et al. in press). These models can rarely be quantified, however, and in many cases the appropriate model for a particular situation may not be apparent before dam removal. Thus future research will need to concentrate on discriminating among the myriad possible geomorphic responses to dam removal and improving the quantitative basis for predictions.

Geomorphic effects of different engineering strategies

The engineering design and implementation of dam removal plans can profoundly influence the subsequent geomorphic evolution of the impoundment, as well as the extent and na-

ALTHOUGH MANY WELL-ESTABLISHED CONCEPTS OF FLUVIAL GEOMORPHOLOGY ARE RELEVANT FOR EVALUATING THE EFFECTS OF DAM REMOVAL, GEOMORPHOLOGISTS REMAIN UNABLE TO FORECAST STREAM CHANNEL CHANGES CAUSED BY THE REMOVAL OF SPECIFIC DAMS

ture of sediment impacts downstream of the dam. Important design considerations include strategies to stabilize or remove sediment fill above the dam, the timing and nature of the actual dam removal, and the extent to which the removal follows engineering design criteria.

A variety of strategies exist for minimizing erosion of the sediment fill above dams. Although it may be expensive, removing sediment fill behind the dam may be useful in some instances (Smith et al. 2000). Removing sediment is a particularly attractive alternative when dam fill sediments present an extreme hazard, or when other exceptional factors can justify the expense involved. Furthermore, the sediment that makes up the fill may consist of sand and gravel (Egan and Pizzuto 2000, Wilcox et al. 2000) that could be sold as aggregate for concrete or for construction fill (assuming that sediments are not contaminated). Regrading, revegetating, and ripraping (i.e., strengthening with a layer of stones) of the exposed dam fill have also been proposed as means of reducing the extent and rate of erosion (Harbor 1993, Kanehl et al. 1997).

Jim Pizzuto (e-mail: pizzuto@udel.edu) received his PhD in geology from the University of Minnesota in 1982. He is a fluvial geomorphologist at the Department of Geology, University of Delaware, Newark, DE 19716. © 2002 American Institute of Biological Sciences.

The engineering design of the actual dam removal may have a significant influence on sediment-related impacts. In many cases, dams are breached only over short sections to allow the reservoir to drain before removing the remainder of the dam (Egan and Pizzuto 2000). Drawing down the reservoir before removal can achieve the same effect, in addition to allowing fine-grained reservoir sediments to consolidate and strengthen (Kanehl et al. 1997). To minimize the potential impacts of eroded reservoir sediments downstream of high dams on the Elwha River in Washington, Harbor (1993) advocated "controlled lowering" of the dams on the Elwha River in Washington, in which removal would occur in stages.

It is also important for engineering designs to be specific and for the removal process to be monitored by qualified inspectors. When the Manatawny Dam in Pottstown, Pennsylvania, was removed in August 2000, the contractor was simply directed to remove the dam: No detailed specifications for removal were given to the contractor, and the dam removal process was not monitored by surveyors. After the dam had been "removed," surveys revealed that the contractor had removed only half of the 2-meter (m) height of this dam. The remaining 1 m of the dam consisted of large blocks that could not be transported by the stream. These blocks made erosion at the dam site impossible, in addition to controlling the elevation of the streambed above the dam. As a result, the channel upstream did not change initially. In November 2000 the remaining half of the dam was removed, and by June 2001 sand and gravel had accumulated in the streambed above the dam and finer-grained sediments had been swept downstream to expose gravel riffles (steep, rocky sections of the channel with shallow, fast-moving water) (Egan 2001).

Such problems may not necessarily be caused by a contractor's negligence. Rather, the desired elevation of the dam site following dam removal may not be explicitly defined or discussed. Indeed, it is an easily neglected concept: What could be simpler than just "removing" the dam? At the Manatawny Dam, the problem was clearly illustrated only following a detailed survey of the longitudinal profile over the dam site. Ideally, the design should have included a target longitudinal profile for the postdam channel, which, when projected over the dam site, would have indicated the appropriate elevation to which the contractor should have excavated.

Geomorphic processes above the dam

Upstream from the dam, geomorphic processes should follow a coherent sequence (figure 1). First, the channel will incise through the sediment fill. Bank failures will occur if the channel depth increases above a critical value that depends on the strength of the soil and the detailed geometry of the stream. The additional sediment supplied by bank failures could be used to build floodplains and, ultimately, a new equilibrium channel. The complete sequence will probably require at least a decade and will depend greatly on the mass and grain size of the sediment stored behind the dam.

Incision processes. The sediment fill in the impoundment could be incised by a variety of processes that will probably depend on the height of the sediment fill and its grain size (figure 2). In cohesive silt and clay sediments, a vertical headcut (an eroding vertical face in the stream bed) is likely to migrate upstream through the fill (Doyle et al. forthcoming). Sandy fills could be subject to sapping as groundwater emerges at the base of a headcut. Other mass wasting processes related to liquefaction of sandy sediment could also occur, particularly when the reservoir fill is thick. Otherwise, a knickpoint (an abrupt increase in slope) could migrate upstream through a sandy fill. Fills composed of sand or cohesive silt and clay are likely to erode even during low flows, but fills composed of gravel may be incised only during high-flow events that are competent to move coarse sediment (Egan 2001, Doyle et al. forthcoming). For this reason, gravel fills are labeled as "event-driven" in figure 2.

Incision rates for removing dam fill sediments are poorly documented. Gerrits (1994) documented 300 m of knickpoint migration in the year following the removal of Musser Dam in Pennsylvania, a 10-m-high run-of-river dam (a small dam that does not significantly influence the water discharge into the stream). Doyle and colleagues (forthcoming) describe the migration of knickpoints following the removal of two low-head dams in Wisconsin, but they do not provide quantitative results.

Development of a stable channel morphology. As noted in considerable detail by Doyle and colleagues (forthcoming), field studies of the development of incised channels provide a useful conceptual model of how channels could respond to dam removal (Schumm et al. 1984, Harvey and Watson 1986, Simon 1989a, 1989b, Simon and Hupp 1992). Harvey and Watson (1986) developed a six-stage conceptual model for the evolution of Oaklimiter Creek in northern Mississippi (also summarized by the Task Committee on River Width Adjustment [TCRWA 1998b]) (figure 3). The six stages may be observed at any time along the longitudinal profile of an incising channel, but they also indicate the evolution of individual cross-sections through time. In stage I, the channel slope is steepened above its equilibrium value, but incision has not yet occurred and the banks are stable. During stage II, the channel incises. Stage III is characterized by extensive bank erosion. The additional supply of sediment from the banks causes aggradation of the bed (stage IV), which gradually becomes vegetated (stage V) and ultimately develops into a mature floodplain with an equilibrium channel (stage VI).

Monitoring studies of incised channels indicate that the complete sequence occurs over several decades. Because comparable observations of dam removals are lacking, the appropriate time scale for incision and recovery following dam removal is undocumented. However, when describing the geomorphic response of the removal of two small low-head dams in Wisconsin, Stanley et al. (2002) observed "relatively small and transient geomorphic changes in downstream

reaches, and apparently rapid channel development to an equilibrium form within the impoundment.” In this case, dam sediments were composed of readily transportable sand, and extensive floodplain development was apparently not required to form an equilibrium channel.

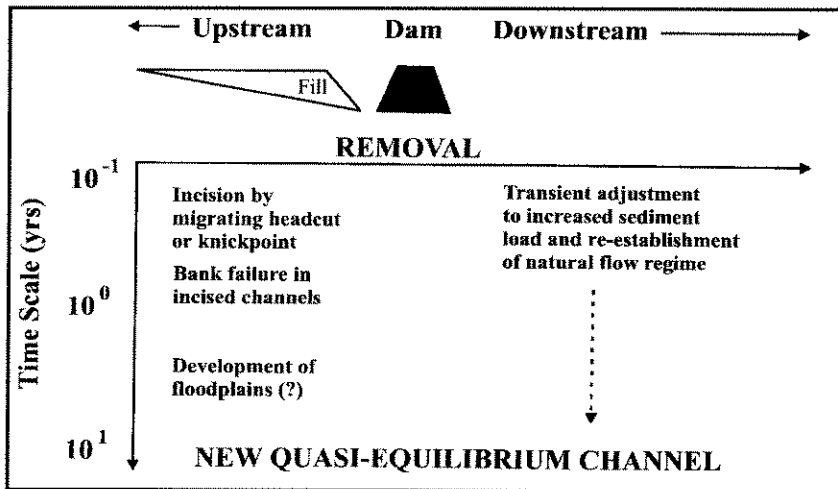


Figure 1. Schematic illustration of geomorphic processes above and below removed dams. The time scale is highly speculative and will vary considerably from site to site, depending on the size of the dam, the mass of sediment impounded, and other variables.

If the sediment fill in the impoundment is thin, incision may not occur (figure 2). Draining a wide impoundment may create extensive flat areas upstream with a wide, shallow channel (Egan and Pizzuto 2000). To develop a narrower, deeper equilibrium channel, floodplains may have to form by vertical accretion as sediment is deposited from overbank flows. These processes are well documented in the geomorphic literature (Schumm and Lichty 1963, Allred and Schmidt 1999, Moody et al. 1999). For example, Moody and colleagues (1999) described floodplain development and channel narrowing following a large flood on the Powder River in southeastern Montana. The floodplain, which grew over approximately 20 years, was built by the deposition of decimeter-thick layers of sand and mud when annual or biannual floods overtopped the growing floodplain.

After the Manatwny Dam was removed, extensive gravel bars formed. These probably represent the initial stages of floodplain development required to narrow the channel by about 10 m (Egan 2001). The deposition required to accrete the gravel bars far exceeds the volume of erosion at Manatwny Dam, indicating that the primary response to dam removal in this case was deposition rather than erosion and incision. Thus sediment budgets for downstream reaches may need to be reconsidered depending on whether incision or floodplain development is expected to dominate at a particular site.

Predicting morphology of the equilibrium channel.

It is often desirable to be able to predict the size and shape of the equilibrium channel that will ultimately form upstream from the dam. Channel width and depth are needed, for

example, to design river restoration projects. However, making such predictions is very difficult. Although the dimensions of undisturbed reaches upstream can provide a useful guide (Egan and Pizzuto 2000, Egan 2001), the banks of the channel within the former impoundment will likely have a different

sediment type and different riparian vegetation from any reach upstream. Thus even empirical methods may not provide an accurate assessment of the equilibrium channel width and depth in the impoundment. Furthermore, empirical methods “cannot predict either the rate of change or intermediate widths attained during dynamic adjustment of channel morphology” (TCRWA 1998b). The American Society of Civil Engineers’ Task Committee on River Width Adjustment provides a useful review of these issues (TCRWA 1998a, 1998b).

Geomorphic processes downstream from the dam

Overview. Downstream from the dam, the channel will respond to the increased sediment load from the eroding fill, as well as to the reestablishment of a natural flow regime.

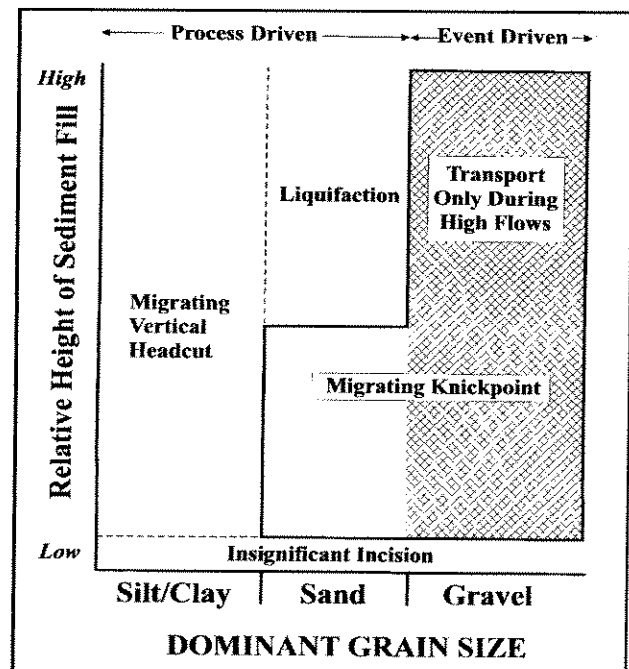


Figure 2. Speculative relationships between the height of a reservoir sediment fill, the dominant grain size of the fill, and different processes of incision. Erosion of gravel depends on high-flow events; therefore these incision processes are “event-driven.” Incision of sand and of silt and clay do not depend on high-flow events, but rather on the mechanism of incision; therefore, removal of fills of sand and of silt and clay are “process-driven.”

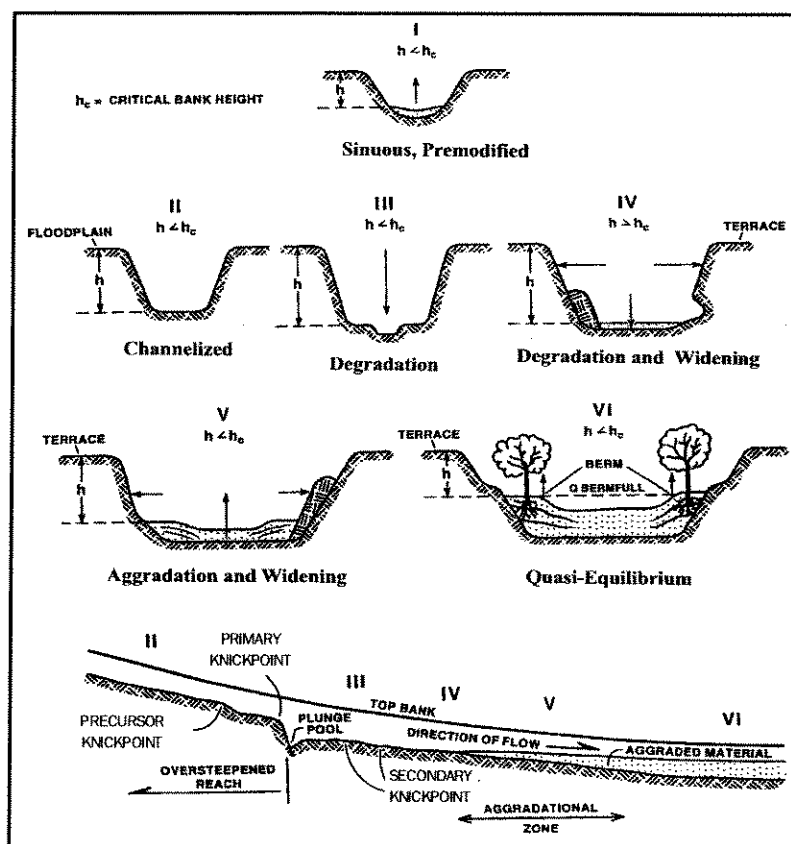


Figure 3. Six-stage sequence of incised channel evolution (from Doyle and Stanley forthcoming, after Simon and Hupp 1992, Harvey and Watson 1986, and TCRWA 1998b). The variable h_c is the critical height at which bank failures occur.

Because the volume of sediment supplied by channel incision will vary with time (Simon 1989), and because channel responses to changes in sediment supply are time-dependent, the morphology and sediment character of the channel downstream will be highly transient. Gradually, as the fill upstream is removed and stored sediment moves downstream, a stable equilibrium form should develop.

Geomorphologists and hydraulic engineers have developed at least two distinct conceptual models to explain patterns of bed elevation changes and sediment transport following a one-time increase in the supply of bed material to a stream channel (figure 4). In one case, the initial pulse of sediment (as represented, for example, by the accumulation of sediment behind a dam) decays in place, a process called dispersion by Lisle et al. (2001). Sediment in transport above the pulse is trapped on its upstream side, and sediment eroded from the crest of the pulse is deposited downstream, resulting in a pattern of bed evolution that resembles a classical diffusion process (e.g., decay of heat from a point source or transport of a dissolved substance by Brownian motion). Dispersion is contrasted with *translation*, in which a wave of bed material travels downstream without a decrease in amplitude. A combination of both processes is also possible (figure 4).

Although many geomorphologists have suggested that sediment inputs translate as waves (Gilbert 1917, Madej and Ozaki 1996), recent experimental (Lisle et al. 1997, 2001), theoretical (Cui and Parker 1997), and field studies (Ball et al. forthcoming) suggest that dispersion should predominate. For example, Lisle et al. (1997) introduced a pulse of sediment into an experimental equilibrium gravel channel. The pulse essentially decayed in place, evolving almost entirely by dispersion (figure 5). Lisle et al. (1997) were able to explain their observations using a relatively simple mathematical model of hydraulics and sediment transport. More extensive flume experiments and modeling results partly reported by Lisle et al. (2001) also emphasize the importance of dispersion. The erosion of a landslide dam on the Navarro River in California also was almost entirely dispersive (Ball et al. forthcoming).

Determining the relative importance of dispersion and translation is significant because the two models have different implications for downstream sediment impacts following dam removal. If a bed material wave translates without decreasing in amplitude, then serious sediment impacts could propagate downstream. Dispersive bed material waves, on the other hand, create sediment impacts that decrease in severity both with time and distance downstream.

Ecological impacts could also vary in response to these two contrasting processes. For example, translation might have larger short-term impacts

at a particular location, but then the sediment wave would pass that location and have no further effect. By contrast, a dispersive process might have a smaller effect per unit time at a particular location, but impacts at that site could last much longer.

The results described above that emphasize the importance of dispersion apply primarily to gravel-bed rivers and do not take into account factors such as floodplain processes and width adjustment. Nonetheless, they suggest that impacts from bed material following dam removal will not influence the channel far downstream. Doyle et al. (forthcoming) and Stanley et al. (2002) disagree, however, and suggest that downstream translation of sediment waves can be significant under certain circumstances following dam removal.

Reach-scale changes in bed texture and morphology. An increase in sediment supply downstream caused by dam removal could have significant impacts at the reach scale, where a reach is defined as a length of stream that contains several pool and riffle sequences or meanders, or that is 10 to 30 channel widths in length (Leopold et al. 1964). These impacts include destruction of pools and riffles, burial of coarse-grained riffles by finer-grained sediment, and modification of bedforms and armor.

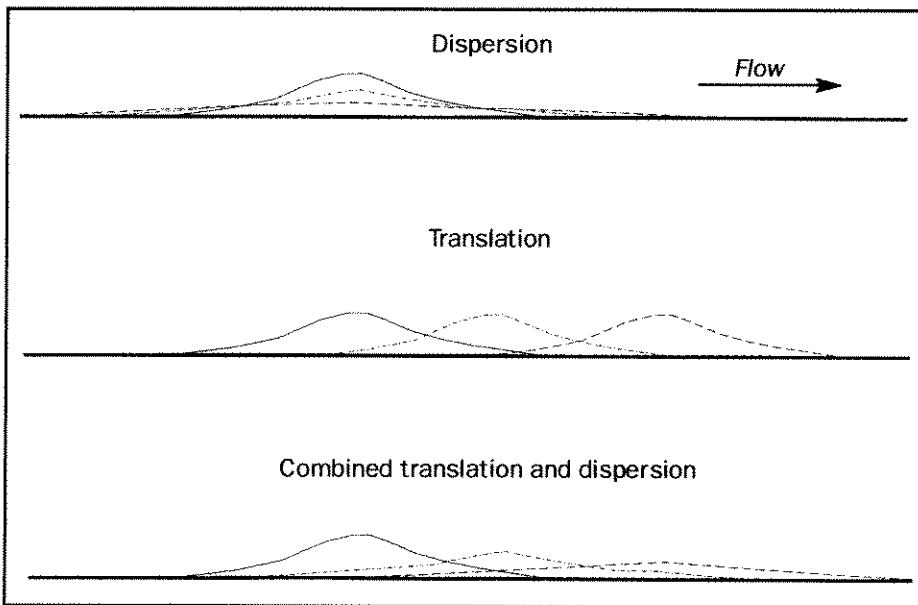


Figure 4. Dispersion, translation, and combined translation and dispersion of bed material waves illustrated in profile. The solid line represents the initial bed topography. The dashed and dotted line represents the second position of the bed, and the dashed line represents the final position of the bed (after Lisle et al. 2001).

dam fill had migrated at least 25 m downstream, destroying both the pools and riffles created by alternate bars and the armored bed. A bed with scattered sandy patches replaced the preexisting armored bed. After 5.2 hours, the armored bed was reestablished, but the alternate bars had not reappeared. Finally, after 8.5 hours, the same pattern of alternate bars and pools and riffles that characterized the initial equilibrium channel had reappeared.

These observations suggest that the sediment supplied by dam removal could rapidly destroy the structure of the bed at the reach scale. This conclusion is supported by many field studies demonstrating a decrease in surface grain size in gravel bed rivers that is caused by an increased supply of finer grained sediment (Montgomery et al. 1999). During the ensuing recovery, as the extra sediment

Flume studies are particularly effective for investigating bed processes in gravel-bed rivers. Analyses of scaling laws and the relevant fluid mechanical principles indicate that small-scale flumes are excellent physical analogues for real gravel-bed rivers (Shvidchenko 1998).

Observations from flume studies suggest that changes in bed texture and morphology resulting from an increase in sediment supply may occur in a predictable sequence (figure 6). The experiments described by Lisle et al. (2001) involved (a) creating an equilibrium channel with an armored gravel bed and well-developed alternate bars (the uppermost map in figure 6), (b) introducing a pulse of sediment that could represent a dam fill (figure 6), and (c) observing the response of the channel downstream as the pulse was eroded. In these experiments, the pulse was approximately 15 channel widths long and 3.5 centimeters (cm) high—about the same height as the equilibrium depth of flow. The transient evolution of the “dam fill” and the reach downstream was observed for 8.5 hours. After 0.6 hours, the sediment from the

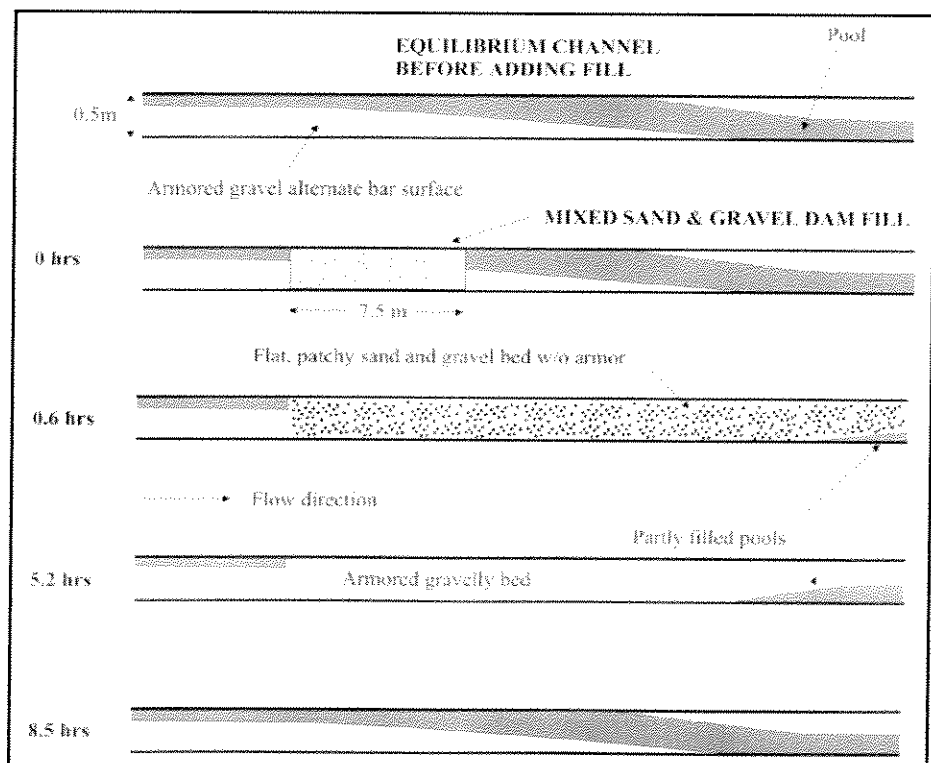


Figure 5. Evolution of a sediment wave in an experimental channel. The horizontal axis represents the longitudinal distance down the flume. The vertical axis is the thickness of sediment above the sloping base of the flume. At 0 hours (hrs), an equilibrium channel is illustrated. After 0.75 hrs, a pulse of sediment 3 cm high and 20 meters long was introduced into the channel. This pulse essentially decayed in place. The “observed” data have been smoothed, and the solid line represents predictions from a mathematical model (after Lisle et al. 1997).

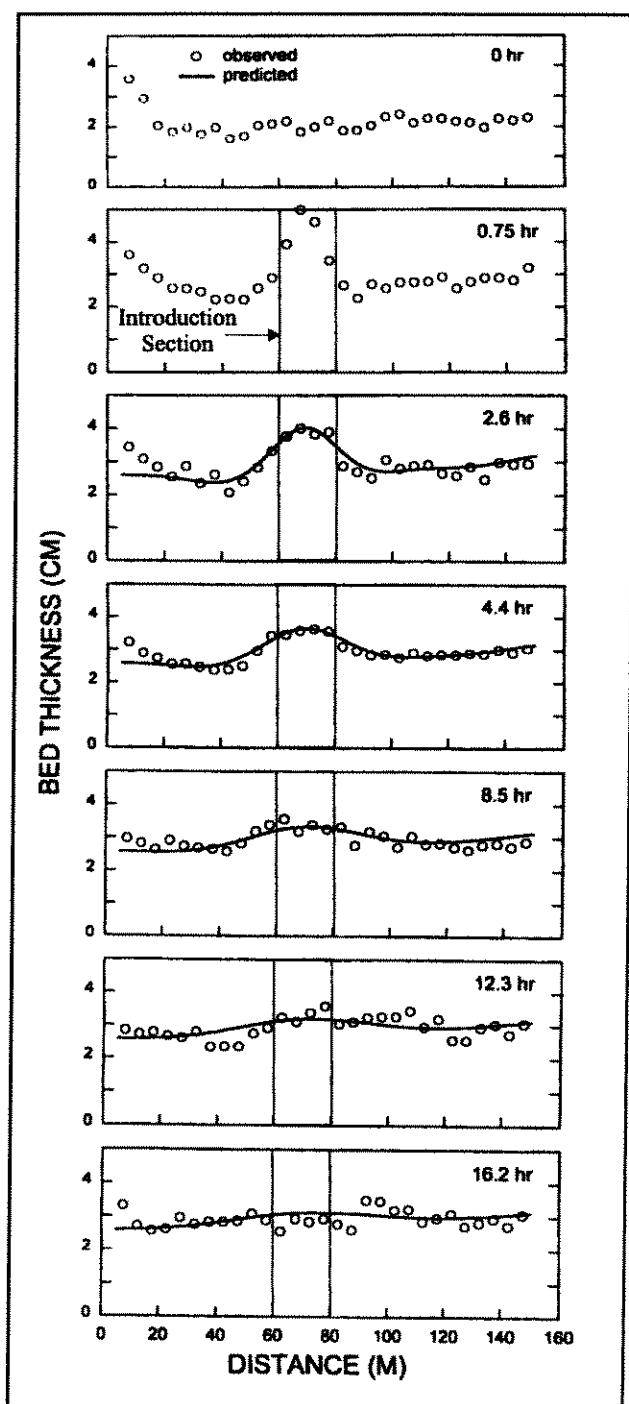


Figure 6. Map views of a laboratory flume experiment modeling the erosion of a sand and gravel dam fill. The uppermost drawing illustrates an equilibrium channel with armored alternate bars. A pulse of sand and gravel (possibly representing a dam fill) was then added; this pulse was equal to the flow depth in height and approximately 15 channel widths long. Three maps illustrate the recovery of the channel to its initial condition after 8.5 hours. These observations were made during run two of experiments at St. Anthony Falls Laboratory by Lisle et al. (2001).

is removed, the armored bed is reestablished first, followed by alternate bars and pools and riffles. Although these changes were observed in a matter of hours in a laboratory flume, the time scale for equivalent changes in a field situation is difficult to specify precisely, but it is likely to be at least several years (Madej 2001).

Observations at the reach scale by Egan (2001) following the removal of Manatawny Dam provided some information on the nature of the evolution of alternate bars and pools and riffles. Downstream from the dam, cobble riffles were buried by a mixture of sand, pebbles, and granules eroded from the dam fill upstream. (Buried riffles following dam removal were also noted by Kanehl et al. 1997 and Stanley et al. forthcoming, documenting aggradation downstream of removed dams.) After 11 months of monitoring, these riffles remained buried. In the impoundment itself, incipient pools and riffles and a midchannel bar formed during a 2.5-year flood (a flood that, on average, will be equaled or exceeded only once every 2.5 years) that occurred 5 months after the dam was removed. After 11 months, however, the spacing of the pools was relatively incoherent compared with a control reach upstream, in which pools and riffles exhibited fairly regular spacing of five channel widths. Pools were also deeper in the control reach than in the impoundment area. These observations suggest that complete development of pools and riffles in a gravel-bed channel following dam removal could take at least several years, depending on the frequency of discharges competent to move the bed sediment.

Other, more complex responses at the reach scale are also possible. Gerrits (1994), for example, documented sediment storage in backwater areas and on the floodplain following the removal of Musser Dam. Stanley and colleagues (2002) observed similar deposits following the removal of small dams in Wisconsin. Sediment could also be stored in areas of low current velocity close to stream banks (Stanley et al. 2002).

Numerical models of geomorphic response

Numerical models are commonly used to evaluate sediment transport and hydraulic processes associated with dam removal. These models predict the average velocity and water surface elevation for a reach, and use these hydraulic data to estimate reach-averaged rates of sediment transport and changes in bed elevation.

At a symposium on "Rehabilitation and Decommissioning of Aging Dams" at the 2000 fall meeting of the American Geophysical Union, five of ten presentations emphasized predictions based on numerical models (the abstracts are published in the 2000 fall meeting supplement of EOS and are also available at www.agu.org). How robust are these predictions?

The impacts of most dam removal projects are likely to extend far enough downstream to require the use of one-dimensional models, rather than more complex two- or three-dimensional models, because of limitations in computer information storage capacity and computational power. For

Table 1. Studies that could lead to improved forecasting of dam removal effects.

Type of study	Benefit
Semi-quantitative observation of many dam removal projects	Develop conceptual models that could be used to forecast controlling processes following dam removal
Improved numerical modeling	Improve quantitative forecasting; assist engineering design
Physical modeling	Test conceptual and numerical models under widely varying controlled conditions
Integrated modeling of geomorphic, hydrologic, and ecological processes	Develop physical models that are useful for predicting biotic and biogeochemical effects
Comprehensive, quantitative, multidisciplinary studies of selected dam removal projects	Rigorous testing of physical and biological predictions.

example, Wilcox et al. (2000) modeled approximately 45 kilometers of the Sandy River in Oregon to predict the potential impacts of the proposed removal of Marmot Dam. Although it would clearly be impractical to represent meter-scale spatial and temporal variations in hydraulics, morphology, and sediment transport over such distances, this is precisely the resolution required by two- or three-dimensional models.

One-dimensional models can predict only changes in grain size and bed elevation in the downstream direction, and all results are averaged across the width of the channel. Furthermore, predictions in the downstream direction are typically associated with a computational grid that is widely spaced relative to channel width. As a result, one-dimensional models predict single, reach-averaged values of grain size and bed elevation. Predictions of smaller scale or multi-dimensional features such as alternate bars or grain-size patches cannot be obtained from one-dimensional models.

One-dimensional sedimentation models are, however, relatively well-established tools in river engineering. Useful reviews of older models were presented by Dawdy and Vanoni (1983) and the National Academy of Sciences (1983). More up-to-date reviews will be available with the publication of the American Society of Civil Engineers Manual 54, *Sedimentation Engineering*, which will contain chapters on "Sediment Transport Mechanics," "Transport of Gravel and Sediment Mixtures," and "1-D Computational Modeling of Sedimentation Processes." One-dimensional models have been used in thousands of field studies (Ball et al. forthcoming provide an excellent example) and laboratory studies (Cui et al. 1996), often with useful results.

Nonetheless, many of the processes represented by current one-dimensional sediment transport models are not well understood. For example, methods for computing transport rates of sand and gravel mixtures are in their infancy (Wilcock 1997). Methods for computing transport processes of silt and clay are also rudimentary (Packman 2001). Mixtures of sand and gravel are very common in nature, and most of the sediment load of rivers is represented by the transport of silt and clay. Furthermore, these processes are only selected examples. Nearly all existing models neglect many other important processes, including upstream propagation of knick-points and headcuts; changes in width due to bank erosion or deposition (TCRWA 1998a, 1998b, Doyle et al. forth-

coming); processes associated with floodplains, including overbank flows and associated sediment transport; and the influence of vegetation on sediment transport processes.

An additional impediment to the development and use of improved numerical models is the poorly developed state of conceptual models that identify controlling geomorphic processes (Grant 2001). As a result, the processes that should be included in a quantitative model forecast at a particular site are not well constrained. Empirical observations are also needed to better define the processes that will probably occur during particular dam removal projects (Grant 2001). It is difficult to provide accurate, quantitative forecasts of the effects of dam removal using a numerical model if the processes represented by the model cannot be identified *before* a dam is removed.

Toward improved forecasting of dam removal effects

Improving our ability to forecast the effects of dam removal will require a concerted, well-designed effort. Table 1 outlines some components of a research program that could help achieve this goal.

Our greatest need is to improve the ability to develop and test conceptual models that will indicate the relevant processes controlling the evolution of the river following dam removal. This will require observations from dam removal projects under a wide variety of conditions, with varying dam heights, fill sediment types, impoundment sizes, and a host of other variables. Because it is impractical to study a large number of dam removal projects in detail, geomorphologists, engineers, ecologists, and others will need to develop rapid protocols for semi-quantitative documentation of dam removal processes through a multidisciplinary effort.

Researchers should also develop improved numerical models to quantify the relevant processes identified by improved conceptual models. Although current models do not include many relevant processes, the rapid development of computing power and the widespread availability of modeling expertise should allow development of useful predictive models. The current widespread use of numerical models indicates that models will always be needed to provide quantitative predictions to guide management decisions. If models are to be used, then both researchers and managers should have confidence in them.

Studies using flumes or other physical models could be extremely useful for improving our conceptual knowledge of dam removal processes and testing numerical models under rigorously controlled conditions. Physical models have provided extremely useful results in many areas of fluvial geomorphology, including landscape evolution (Hasbargen and Paola 2000), the development of drainage basins (Parker 1976), watershed-scale sediment routing (Parker 1976), the evolution of localized sediment inputs (Lisle et al. 1997, 2001), the development of armor in gravel bed rivers (Parker et al. 1982), and the evolution of bedrock channels (Wohl and Ikeda 1997).

Physical models could provide a cost-effective means of studying dam removal processes under controlled conditions that cannot be duplicated by field studies. Scaled physical models have significant limitations, however. Sedimentary processes involving clay, silt, or fine sand often cannot be effectively scaled because of the surface chemistry of the finest grain sizes. Varying discharges are difficult to create and scale in the laboratory, and scale models cannot represent important effects caused by vegetation in the field. Finally, the geometry of scale models does not always correspond to field conditions.

Developing improved conceptual and numerical models of hydrodynamic and geomorphic processes will not suffice. Ecologists also need predictions of changing river morphology and sediment transport processes to predict changes in ecological processes following dam removal. However, the nature and scale of geomorphic predictions that are most useful to ecologists are not necessarily those that geomorphologists are most likely to produce. For example, a one-dimensional model used by an engineer or geomorphologist might predict the mean grain size of the bed material or even the extent of bed armoring. Ecological processes might be linked more strongly to the percentage of silt and clay in the bed—a quantity that has received little attention from geomorphologists and engineers. To maximize the utility of geomorphic predictions for ecologists, a coordinated multidisciplinary effort is needed to develop integrated geomorphic and ecological models. This will require a conscious, planned collaboration between ecologists and geomorphologists throughout entire projects, from initial study design to final model development and testing.

To gain confidence in the reliability and precision of improved predictive models, comprehensive, quantitative, multidisciplinary monitoring studies are needed. A coordinated study of the removal of a dam on Manatawny Creek in Pottstown, Pennsylvania, provides a useful example (Johnson 2001). These studies will be expensive and difficult, and therefore only a small number of such efforts can be funded. However, they represent the only means of thoroughly evaluating our forecasting ability and of understanding the effects of dam removal on fluvial and biological processes.

Conclusions

Previous research on fluvial processes provides many useful models for evaluating the geomorphic effects of dam removal. Studies of the evolution of incised channels, knick-point and headcut migration, floodplain formation and channel narrowing, bank erosion and channel widening, the movement of sediment waves, the formation of alternate bars, the origin of patches of differing grain sizes in gravel bed rivers, and the development of armored beds all provide insights into potential trajectories of channel evolution following dam removal.

Upstream from the dam, geomorphic processes should be dominated by evolution of the channel as it incises into the sediments trapped in the impoundment. Case studies of the evolution of incised channels suggest several stages that will ultimately lead to development of a new equilibrium channel. The initial stages involve downcutting, followed by bank erosion and aggradation of the bed and floodplain development. If the impoundment contains relatively little sediment and is significantly wider than equilibrium channels upstream and downstream of the dam, then the primary processes above the dam are likely to be deposition and floodplain construction (Egan 2001) rather than erosion and incision.

Downstream from the dam, geomorphic processes should be dominated by fluvial responses to temporally varying sediment supply. Observations in the field and in laboratory flumes suggest that the dam fill will not migrate downstream as a coherent "sediment wave," but is more likely to disperse in place, leading to sediment impacts that decrease with the time since removal and the distance from the dam. Increased sediment supply at the reach scale could destroy alternate bars, pools and riffles, and armored beds. Enhanced sediment storage on floodplains, in backwater areas, and along the banks is also likely. The time scale for recovery from downstream transient sediment impacts is currently difficult to predict, but the available evidence suggests that years or decades may be required.

Although a variety of useful models exist for predicting the geomorphic effects of dam removal, site-specific forecasts are unlikely to be reliable. Coordinated research is needed to define the geomorphic processes that are most likely to dominate under different conditions, develop improved conceptual and numerical models, couple geomorphic and ecological models, and monitor selected dam removal projects in sufficient detail to evaluate both qualitative and quantitative forecasts.

The geomorphic effects of dam removal can be significantly influenced by different strategies of design, management, and construction. The removal process can potentially be scheduled and manipulated to minimize undesirable impacts. A variety of methods are available to control erosion of the sediment fill and therefore to minimize the effects of increased sediment supply downstream. Well-conceived restoration strategies could potentially increase the rate of recovery both above and below the dam. Future research programs

should be designed to provide the scientific knowledge to guide management decisions so that informed choices can be made as to whether dams should be removed, and if so, how, when, and where.

Acknowledgments

This paper would not have been written without Dave Hart's encouragement and enthusiasm. Helpful reviews were provided by Dave Hart and Thomas Johnson of the Patrick Center of the Academy of Natural Sciences and three anonymous reviewers. I would particularly like to thank The Pew Charitable Trusts for providing support that helped me complete this paper.

References cited

- Allred TM, Schmidt JC. 1999. Channel narrowing by vertical accretion along the Green River near Green River, Utah. *Geological Society of America Bulletin* 111: 1757–1772.
- Ball MH, Sutherland DG, Hilton SJ, Lisle TE. Evolution of a landslide-induced sediment wave in the Navarro River, California. *Geological Society of America Bulletin*. Forthcoming
- Collier M, Webb RH, Schmidt JC. 1997. Dams and Rivers: Primer on the Downstream Effects of Dams. Tucson (AZ): US Geological Survey.
- Cui Y, Parker G. 1997. Linear analysis of coupled equations for sediment transport. Pages 1256–1261 in 27th International Congress. San Francisco: International Association of Hydraulic Research.
- Cui Y, Parker G, Paola C. 1996. Numerical simulation of aggradation and downstream fining. *Journal of Hydraulic Research* 34: 185–204.
- Dawdy D, Vanoni V. 1983. Modeling alluvial channels. *Water Resources Research* 22: 715–815. Forthcoming.
- Doyle MW, Stanley EH, Harbor JM. Geomorphic analogies for assessing probable channel response to dam removal. *Journal of the American Water Resources Association*. Forthcoming.
- Egan J. 2001. Geomorphic effects of dam removal on the Manatawny Creek, Pottstown, PA. Master's thesis, Department of Geology, University of Delaware, Newark.
- Egan J, Pizzuto JE. 2000. Geomorphic effects of the removal of the Manatawny Dam, Pottstown, PA. EOS, Transactions, American Geophysical Union 81 (fall meeting supplement).
- Gerrits M. 1994. Physical effects on Middle Creek due to the decommission of Musser Dam. Geology senior thesis, Bucknell University, Lewisburg, PA.
- Gilbert GK. 1917. Hydraulic Mining Debris in the Sierra Nevada. Washington (DC): US Geological Survey.
- Grant G. 2001. Dam removal: Panacea or Pandora for rivers? *Hydrological Processes* 15: 1531–1532.
- Harbor JM, ed. 1993. Proposed Measures to Alleviate the Environmental Impacts of Hydroelectric Dams on the Elwha River, Washington, U.S.A. Alexandria (VA): Water Environment Federation.
- Harvey MD, Watson CC. 1986. Fluvial processes and morphological thresholds in incised channel restoration. *Water Resources Bulletin* 3: 359–368.
- Hasbargen LE, Paola C. 2000. Landscape instability in an experimental drainage basin. *Geology* 28: 1067–1070.
- Johnson TE, Pizzuto J, Egan J, Bushaw-Newton K, Hart D, Lawrence J, Lynch E. 2001. The Manatawny Creek dam removal: Project overview and geomorphic characteristics. *Bulletin of the North American Benthological Society* 18: 121–122.
- Kanehl PD, Lyons J, Nelson JE. 1997. Changes in the habitat and fish community of the Milwaukee River, Wisconsin, following removal of the Woolen Mills Dam. *North American Journal of Fisheries Management* 17: 387–400.
- Lisle TE, Cui Y, Parker G, Pizzuto JE, Dodd AM. 2001. The dominance of dispersion in the evolution of bed material waves in gravel-bed rivers. *Earth Surface Processes and Landforms* 26: 1409–1420.
- Lisle TE, Pizzuto JE, Ikeda H, Iseya F, Kodama Y. 1997. Evolution of a sediment wave in an experimental channel. *Water Resources Research* 33: 1971–1981.
- Madej MA, Ozaki V. 1996. Channel response to sediment wave propagation and movement, Redwood Creek, California. *Earth Surface Processes and Landforms* 21: 911–927.
- Montgomery DR, Panfil MS, Hayes SK. 1999. Channel-bed mobility response to extreme sediment loading at Mount Pinatubo. *Geology* 27: 271–274.
- Moody JA, Pizzuto JE, Meade RH. 1999. Ontogeny of a flood plain. *Geological Society of America Bulletin* 111: 291–303.
- National Academy of Sciences, Committee on Hydrodynamic Computer Models for Flood Insurance Studies, Advisory Board on the Built Environment. 1983. An Evaluation of Flood-Level Prediction Using Alluvial River Models. Washington (DC): National Research Council.
- Packman AL. 2001. Re-examination of the wash load concept: Role of physicochemical processes. *World Water and Environmental Resources Congress*. Orlando (FL): American Society of Civil Engineers.
- Parker G, Dhamotharan S, Stefan H. 1982. Model experiments on mobile, paved gravel bed streams. *Water Resources Research* 18: 1395–1408.
- Parker RS. 1976. Experimental Study of Drainage Basin Evolution and its Hydrologic Implications. Fort Collins: Colorado State University.
- Schumm SA, Harvey MD, Watson CC. 1984. Incised Channels: Morphology, Dynamics, and Control. Littleton (CO): Water Resources Publications.
- Shvidchenko AB, Kopalina ZD. 1998. Hydraulic modeling of bed load transport in Gravel-bed Laba River. *Journal of Hydraulic Engineering* 124: 778–785.
- Simon A. 1989a. The discharge of sediment in channelized alluvial streams. *Water Resources Bulletin* 25: 1177–1188.
- . 1989b. A model of channel response in disturbed alluvial channels. *Earth Surface Processes and Landforms* 14: 11–26.
- Simon A, Hupp C. 1992. Geomorphic and vegetative recovery processes along modified stream channels of West Tennessee. Washington (DC): US Geological Survey.
- Smith IW, Dittmer E, Prevost M, Burt DR. 2000. Breaching of a small irrigation dam in Oregon: A case history. *North American Journal of Fisheries Management* 20: 205–219.
- Stanley EH, Luebke MA, Doyle MW, Marshall DW. 2002. Short-term changes in channel form and macroinvertebrate communities following low-head dam removal. *Journal of the North American Benthological Society* 21: 172–187.
- [TCRWA] Task Committee on River Width Adjustment, American Society of Civil Engineers. 1998a. River width adjustment, II: Modeling. *Journal of Hydraulic Engineering* 124: 903–917.
- . 1998b. River width adjustment, I: Processes and mechanisms. *Journal of Hydraulic Engineering* 124: 881–902.
- Wilcock PR. 1997. A method for predicting sediment transport in gravel-bed rivers. Baltimore: Johns Hopkins University.
- Wilcox AC, Cui Y, Vick JC. 2000. Geomorphic assessment and numerical modeling of sediment transport associated with dam removal: Case study of Marmot Dam, Sandy River, OR. EOS, Transactions, American Geophysical Union 81 (fall meeting supplement).
- Williams GP, Wolman MG. 1984. Downstream effects of dams. Washington (DC): US Geological Survey.
- Wohl E, Ikeda H. 1997. Experimental simulation of channel incision into a cohesive substrate at varying gradients. *Geology* 25: 295–298.

A Special Section on Dam Removal and River Restoration

DAVID D. HART AND N. LEROY POFF

Human activities have degraded many of the world's ecosystems, which has created an urgent need for strategies that can restore their ecological integrity. This need is accompanied by many scientific challenges, however. In particular, ecosystems are among the most complex entities in the hierarchy of life, and the successful repair of damaged systems will require a deep understanding of the processes that determine their structure and function. Biologists have a critical role to play in creating this knowledge because of their expertise in such varied phenomena as the role of microbes in detoxifying anthropogenic contaminants, the effects of disturbance on population persistence, and the factors influencing competitive interactions between native and exotic species.

By itself, however, biological knowledge is not sufficient for restoring degraded ecosystems. Two other types of expertise are also needed for developing integrated restoration solutions. First, because ecosystems are composed of many interacting abiotic and biotic components, biologists must collaborate with their colleagues in the physical sciences to learn how these systems work. Second, because humans are such strong interactors in these complex systems, we need to work with experts who can help us understand how human attitudes, institutions, and technologies influence the condition and management of ecosystems. Such enhanced interdisciplinary dialogue and unified approaches are essential for creating public policies that can sustain the planet's life support systems (Lubchenco 1998, Covich 2000, Ludwig et al. 2001).

Proposals to restore rivers via dam removal raise many issues that require broad discussion and teamwork. This approach to river restoration derives from the growing recognition that dams often disrupt the structure and function of river ecosystems by modifying flow regimes, disrupting sediment transport, altering water quality, and severing their biological continuity (Ward and Stanford 1979, Petts 1984, Collier et al. 1996). Future dam removal decisions can be

enhanced by developing a more complete scientific understanding of the processes that determine how rivers are affected by different types of dams and how they respond to dam removal. There is an equally important need to understand the social, economic, engineering, and legal factors that influence dam removal decisions. Assembling a diverse array of experts to explore these different facets of dam removal was an exciting challenge for us. Listening to and participating in the dialogue that took place when those experts gathered at the annual meeting of the Ecological Society of America in August 2001 was even more rewarding.

This special section of *BioScience* brings together those diverse authorities, and a few others, to examine the potential utility of dam removal as a method of river restoration. Our goal is not just to explore the many different scientific and social aspects of this topic but also to consider how these components can and should be connected. Bruce Babbitt, former secretary of the US Department of the Interior during the Clinton administration, is intimately familiar with the subject matter, having been present—sledgehammer in hand—at many dam removals across the United States. His passionate essay (Babbitt 2002) clearly frames both the scientific and human dimensions of the subject. In particular, he emphasizes the critical need for strong science, not just to predict what will happen when dams are removed but also to monitor dam removal outcomes so that we learn how to maximize the effectiveness of this restoration method.

David D. Hart (e-mail: hart@acnatsci.org) is an ecologist, academy vice president, and director of the Patrick Center for Environmental Research at the Academy of Natural Sciences, 1900 Benjamin Franklin Parkway, Philadelphia, PA 19103; N. LeRoy Poff (e-mail: poff@lamar.colostate.edu) is an associate professor in the Department of Biology at Colorado State University in Fort Collins, where he teaches and conducts research in riverine ecology. © 2002 American Institute of Biological Sciences.

The next six articles focus on various scientific facets of dam removal and river restoration. Poff and Hart (2002) provide an overview of the ways in which dams impair river ecosystems, and they highlight the conceptual and data needs for creating a more systematic and robust science of dam removal. They quantify variation in such important dam characteristics as size and operation at a national scale and describe how this variation can form the basis of an ecological classification system that distinguishes the environmental effects of different dam types. Hart and colleagues (2002) review alternative methods for predicting ecological responses to dam removal, emphasizing that knowledge of dam effects and removal responses is far from complete. They develop and have begun to implement an ecological risk assessment framework for determining how potential effects of dam removal vary as a function of dam and river attributes.

Dam removal can cause dramatic changes in fluvial processes and channel morphology, which will in turn affect many other ecosystem components. Pizzuto (2002) examines the challenges involved in predicting the effects of dam removal on sediment transport and channel evolution; he also suggests a range of studies (including collaborations between geomorphologists, engineers, and ecologists) that could lead to improved forecasting. One example of the benefits of such collaboration is the work by Stanley and Doyle (2002), which examines links between geomorphic processes and nutrient dynamics. They describe how nutrient cycling is influenced by various impoundment processes (e.g., sedimentation, denitrification) and show how geomorphic models can help predict changes in nutrient retention after dam removal.

The removal of very large (> 30 meters high) dams has been proposed as a method for restoring endangered anadromous salmon in the Pacific Northwest, but no dams of this size have yet been removed in the United States. Gregory and colleagues (2002) focus particular attention on the complex web of direct and indirect pathways by which large dams modify ecological interactions in major rivers. They illustrate many of the scientific uncertainties associated with large dam removal through case studies of dams in the Elwha and Snake Rivers, and they explore various options for making prudent decisions in the face of such uncertainty. The removal of dams affects not only aquatic biota, but also the riparian habitats associated with river margins and floodplains. Shafroth and colleagues (2002) examine how riparian vegetation is likely to respond to various geomorphic and hydrologic changes stemming from dam removal; they also discuss how sediment management and vegetation planting strategies can be used to enhance restoration outcomes.

The final three articles focus on the economic, social, and legal dimensions of dam removal. Cost-benefit analysis has been proposed as an important economic tool for evaluating the potential consequences of dam removal. Whitelaw and MacMullan (2002) present a conceptual framework for estimating the costs and benefits of dam removal and examine the way such analyses have been performed for dams on the Lower Snake River. They argue for a balanced approach to

cost-benefit analysis, one that accounts for all subsidies and externalities and places both costs and benefits in a realistic economic context.

Economic issues are not necessarily the primary determinant of stakeholder attitudes and behaviors regarding dam removal, however. For example, Johnson and Graber (2002) have found that communities are often reluctant to consider the removal of old and obsolete dams, even when removal costs much less than dam repair. They describe some of the social and psychological barriers that prevent individuals and communities from considering dam removal as an option and propose creative methods (e.g., community-based social marketing, diffusion of innovations) for encouraging the adoption of management practices that can restore river ecosystems.

Environmental laws might also be expected to provide a powerful tool for removing dams that impair river ecosystems, but Bowman (2002) shows how laws designed to protect ecosystems can actually be an impediment to ecological restoration. Specifically, environmental laws are often designed to protect the environment by maintaining the status quo (i.e., by preventing degradation), which can inadvertently discourage restoration activities because they also cause a deviation (albeit positive) from the status quo. Bowman suggests that regulatory modifications within existing laws might provide decisionmakers with greater flexibility to approve projects with restoration objectives, although she emphasizes that project outcomes must be assessed carefully to avoid creating loopholes that result in environmental degradation.

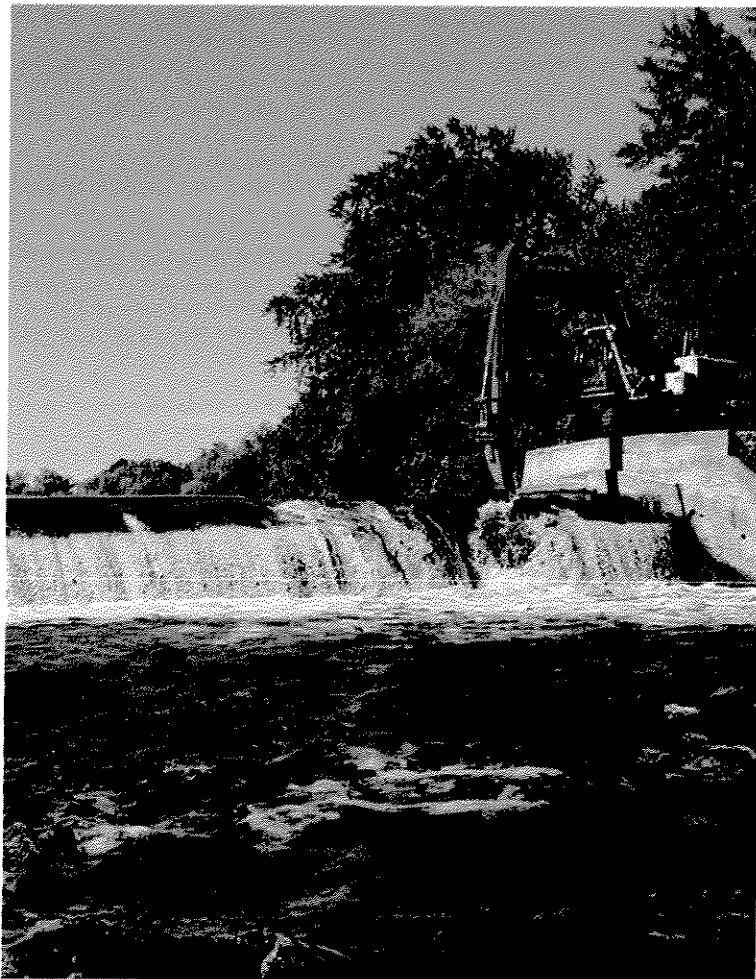
Ultimately, the benefits of this collection of articles may be twofold. First, we hope that it succeeds in calling attention to the potential utility of dam removal in restoring rivers and in focusing research on specific scientific, engineering, and socioeconomic questions that can enhance the effectiveness of this innovative restoration method. Second, it may highlight the need for greater dialogue and closer interaction among a diverse array of experts and stakeholders. Many environmental problems would benefit from broader discourse about the best ways to create sound environmental policies, effective management practices, and adaptive institutions that can restore and protect the ecosystems on which all life depends.

Acknowledgments

We thank the authors for their dedication and enthusiasm, Matt Greenstone for his broad perspective and unflagging optimism, and Robin Davis for capable editorial assistance. We are also grateful to the Aquatic Section of the Ecological Society of America and its officers (Steve Kohler, Craig Osenberg, and Margaret Palmer) for sponsoring our symposium ("Dam removal and river restoration: Linking scientific, socioeconomic, and legal perspectives") at the 2001 annual meeting of the Ecological Society of America. The Pew Charitable Trusts provided support to help participants attend the symposium and prepare manuscripts for publication.

References cited

- Babbitt B. 2002. What goes up, may come down. *BioScience* 52: 656–658.
- Bowman MB. 2002. Legal perspectives on dam removal. *BioScience* 52: 739–747.
- Collier M, Webb RH, Schmidt JC. 1996. Dams and rivers: Primer on the downstream effects of dams. Reston (VA): US Geological Survey. Circular 1126.
- Covich A. 2000. Biocomplexity and the future: The need to unite disciplines. *BioScience* 50: 1035.
- Gregory S, Li H, Li J. 2002. The conceptual basis for ecological responses to dam removal. *BioScience* 52: 713–723.
- Hart DD, Johnson TE, Bushaw-Newton KL, Horwitz RJ, Bednarek AT, Charles DF, Kreeger DA, Velinsky DJ. 2002. Dam removal: Challenges and opportunities for ecological research and river restoration. *BioScience* 52: 669–681.
- Johnson SE, Graber BE. 2002. Enlisting the social sciences in decisions about dam removal. *BioScience* 52: 731–738.
- Lubchenco J. 1998. Entering the century of the environment: A new social contract for science. *Science* 279: 491–497.
- Ludwig D, Mangel M, Haddad B. 2001. Ecology, conservation, and public policy. *Annual Review of Ecology and Systematics* 32: 481–517.
- Petts GE. 1984. Impounded rivers: Perspectives for ecological management. New York: John Wiley and Sons.
- Pizzuto JE. 2002. Effects of dam removal on river form and process. *BioScience* 52: 683–691.
- Poff NL, Hart DD. 2002. How dams vary and why it matters for the emerging science of dam removal. *BioScience* 52: 659–668.
- Shafroth PB, Friedman JM, Auble GT, Scott ML, Braatne JH. 2002. Potential responses of riparian vegetation to dam removal. *BioScience* 52: 703–712.
- Stanley EH, Doyle MW. 2002. A geomorphic perspective on nutrient retention following dam removal. *BioScience* 52: 693–701.
- Ward JV, Stanford JA, eds. 1979. *The Ecology of Regulated Streams*. New York: Plenum Press.
- Whitelaw E, MacMullan E. 2002. A framework for estimating the costs and benefits of dam removal. *BioScience* 52: 724–730.



Potential Responses of Riparian Vegetation to Dam Removal

PATRICK B. SHAFROTH, JONATHAN M. FRIEDMAN, GREGOR T. AUBLE, MICHAEL L. SCOTT, AND JEFFREY H. BRAATNE

Throughout the world, riparian habitats have been dramatically modified from their natural condition. Dams are one of the principal causes of these changes, because of their alteration of water and sediment regimes (Nilsson and Berggren 2000). Because of the array of ecological goods and services provided by natural riparian ecosystems (Naiman and Decamps 1997), their conservation and restoration have become the focus of many land and water managers. Efforts to restore riparian habitats and other riverine ecosystems have included the management of flow releases downstream of dams to more closely mimic natural flows (Poff et al. 1997), but dam removal has received little attention as a possible approach to riparian restoration.

The riparian vegetation that grows in post-dam removal environments interacts strongly with other factors that are generally given more direct consideration in dam removal efforts. For example, riparian vegetation can stabilize sediments in former reservoir pools, perhaps reducing downstream sediment transport that can harm aquatic ecosystems (Bednarek 2001). Vegetation that occupies new surfaces downstream and within the former reservoir pool will influence use by wildlife and for human recreation (AR/FE/TU 1999).

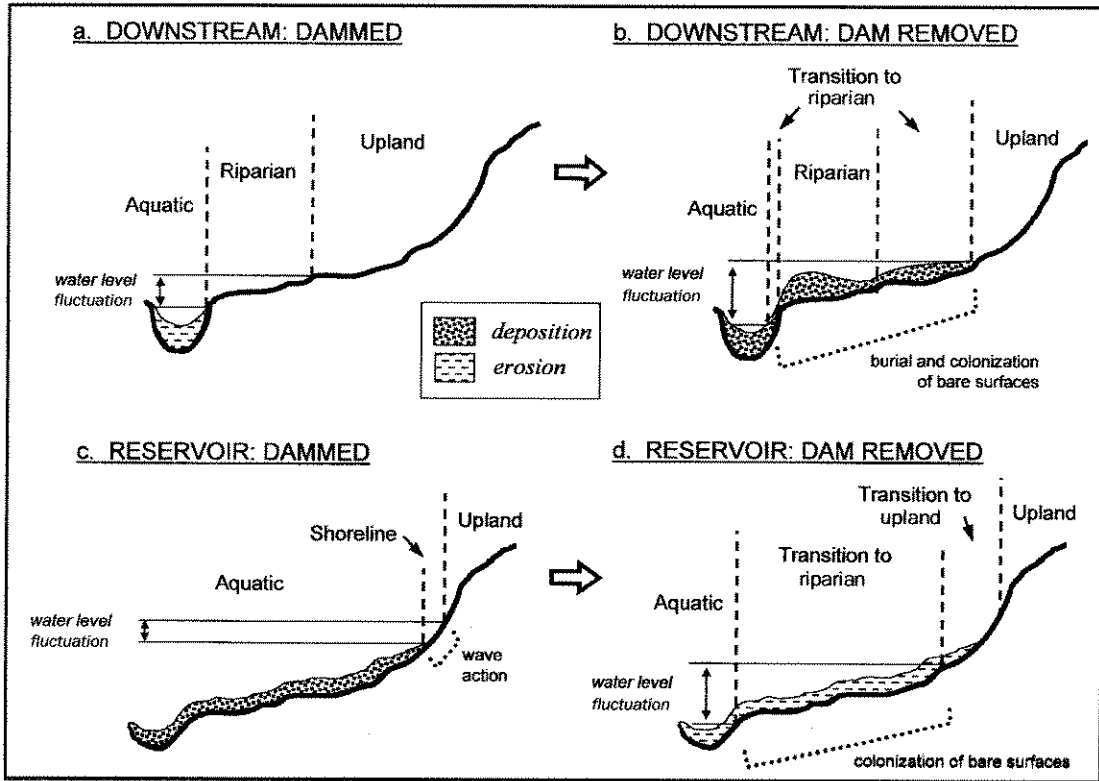
Vegetation response to dam removal is highly dependent on changes to the physical environment. Vegetation at the interface between a water body and the surrounding uplands is dominantly structured by the hydrologic gradient. Sites along this gradient differ in the duration, frequency, and timing of inundation (generally referred to as *hydroperiod*). Species differences in hydroperiod tolerances and requirements produce zonation and pattern in species composition and general cover types along the hydrologic gradient (figure 1). Dam removal may change aspects of the hydrological regime that structure riparian vegetation, including flood and low-flow regimes and associated water table dynamics. Further, dam removal will generally result in the creation of two classes of bare sediment that can be colonized by riparian

DAM REMOVAL GENERALLY CAUSES CHANGES TO ASPECTS OF THE PHYSICAL ENVIRONMENT THAT INFLUENCE THE ESTABLISHMENT AND GROWTH OF RIPARIAN VEGETATION

plants: (1) downstream deposits transported from the former reservoir pool and upstream sources and (2) surfaces within the former reservoir pool (figure 1).

The distribution and character of new bare substrates will vary tremendously across sites. Removal of small dams in systems with low sediment transport may result in few downstream changes and relatively simple upstream changes associated with vegetation colonization and succession on the former lake bottom. Removal of dams that have trapped large quantities of sediment could result in erosion of those deposits and transport of sediment downstream. The physical (e.g., particle-size distribution) and chemical (e.g., macronutrient and micronutrient status) character of sediments may be different from conditions that existed before dam removal, potentially affecting species composition of

Patrick B. Shafroth (e-mail: Pat_Shafroth@usgs.gov) is a plant ecologist, Jonathan M. Friedman is a fluvial geomorphologist, Gregor T. Auble is an ecologist, and Michael L. Scott is a plant ecologist with the US Geological Survey's Fort Collins Science Center, Fort Collins, CO 80526-8118. They conduct research on riparian vegetation in western North America. Jeffrey H. Braatne is an assistant professor of riparian ecology, Department of Fish, Wildlife, and Range Resources, College of Natural Resources, University of Idaho, Moscow, ID 83844. His research focuses on the life history and ecology of native willows and cottonwoods and the impacts of grazing and flow regulation on riparian vegetation.



viable for between 45 and 400 years (Leck 1989). Vegetative reproduction can also be an important strategy for expansion of remnant or founder populations (Krasny et al. 1988, Kalliola et al. 1991).

In this article, we review the scant information documenting responses of terrestrial vegetation to dam removal and derive expected responses both upstream and downstream of the former dam on the basis of empirical and theoretical relationships between riparian plants, stream hydrology, and fluvial processes. We evaluate case studies from North America of planned or completed dam removals, natural analogs of dam removal, and alternative strategies of releasing and exposing water and sediment. We consider transient and equilibrium responses and the effects of different dam removal strategies on native and exotic plants. We focus on the natural establishment of vegetation following dam removal, although we also discuss active measures such as planting.

Downstream responses

Effects of a downstream sediment pulse.

Dams generally trap and store sediment, often depleting reaches downstream (Williams and Wolman 1984). Dam removal may result in the downstream transport of stored sediment, which is usually seen as a potential problem (Simons and Simons 1991, Hotchkiss et al. 2001). For example, the sediment may kill fish, clog spawning gravels, or damage neighboring property. However, this transient pulse of sediment provides an opportunity for channel change and the creation of new surfaces suitable for the reproduction of riparian pioneer species (figures 1, 2a). Such surfaces may have been scarce following dam construction; thus, from the perspective of riparian vegetation, sediment released upon dam removal may be a benefit (Semmens and Osterkamp 2001).

Most dam removals to date have involved small reservoirs with small amounts of sediment, and few data are available concerning the effects of the downstream pulses of sediment on channel morphology and vegetation (Hotchkiss et al. 2001). There are, however, better-described cases of sedi-

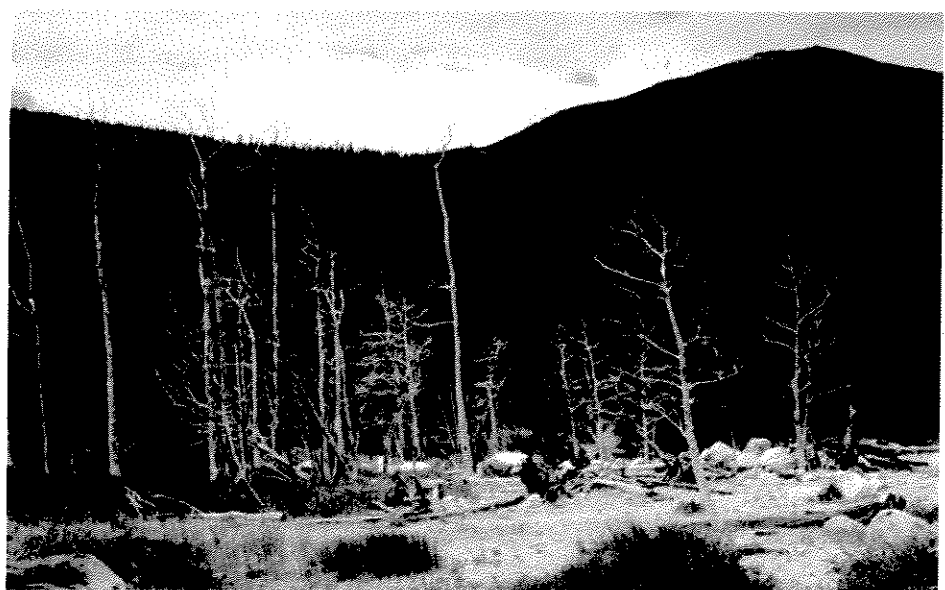
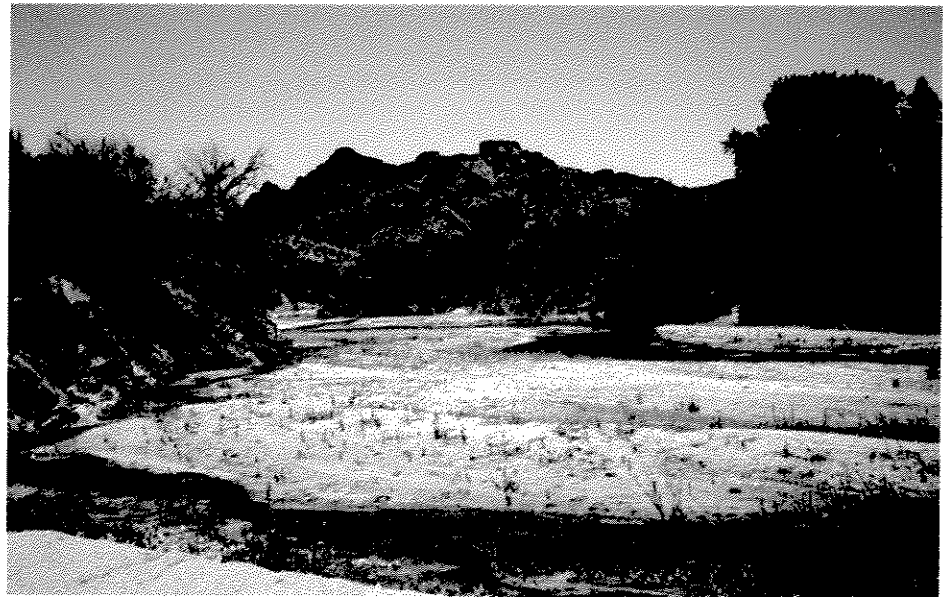


Figure 2. (a) Pioneer riparian vegetation colonizing a new sediment deposit. Fresh alluvial deposits such as these would be expected to occur on river reaches downstream of a dam removal. (b) Tree mortality associated with burial by sediment transported and deposited following a dam failure in Rocky Mountain National Park, Colorado. Photographs by Patrick Shafroth.

ment pulses resulting from other causes, including hydraulic mining (Gilbert 1917, James 1989), timber cutting (Madej and Ozaki 1996), volcanic eruption (Major et al. 2000), large floods (Jarrett and Costa 1993), and dam maintenance (Wohl and Cenderelli 2000). Several generalizations may be drawn from this literature. As the sediment pulse travels downstream, its amplitude decreases and its wavelength increases over time (Gilbert 1917, Simons and Simons 1991, Pizzuto 2002). At a point along the stream, the pulse may be observed as a transient increase in bed elevation or in the rate of sediment transport. Because fine particles are transported more easily than coarse particles, the sediment pulse may be

sorted over time, with finer particles moving downstream more rapidly. The trailing limb of this pulse can take the form of exponential decay, and it may take decades for sediment loads to return to prepulse conditions (James 1989, Simons and Simons 1991). The sediment pulse may partially or completely fill channels, resulting in temporary or permanent channel avulsion. Avulsion and fluctuations in bed elevation often leave behind terrace deposits (James 1989) that may persist for centuries or more. Vegetation may colonize these terrace deposits, as with some valley oak (*Quercus lobata*) forests in California's Central Valley. Other surfaces associated with temporally and spatially variable aggradation and degradation of the sediment pulse will be colonized by vegetation, as has been described for mudflows associated with volcanic eruption (Halpern and Harmon 1983).

In addition to creating new alluvial surfaces, sediment deposition downstream of a removed dam could bury existing vegetation (figure 2b). Riparian species vary in their tolerance of high sedimentation rates (Hupp 1988). If vegetation downstream of dams has succeeded to late seral stages (Johnson 1992), then dominant species in these communities are likely to be less tolerant than pioneering species of burial by sediment. In 1982, a dam breach in Rocky Mountain National Park resulted in a large flood that deposited a 0.18 square-kilometer (km²) alluvial fan that was up to 13.4 meters (m) thick (average thickness = 1.6 m; Jarrett and Costa 1993). Some vegetation died immediately because of complete burial (Keigley 1993), while many trees succumbed over a period of years, probably because of the effects of anoxic soils and accumulations of toxic levels of micronutrients (figure 2b; Barrick and Noble 1993).

Effects of a naturalized downstream flow regime.

Along rivers, the hydrologic regime interacts strongly with the geomorphic setting to influence the establishment and growth of riparian plants. Dam removal could restore natural hydrologic regimes, which can contribute to the rehabilitation of native plant communities (Poff et al. 1997, Taylor et al. 1999, Stromberg 2001). Regulated flow regimes are generally less variable than unregulated flows, and some vegetation downstream of dams is more competitive under relatively homogeneous flow regimes. The timing, magnitude, and duration of flood, flood recession, and baseflows strongly influence riparian vegetation (Rood et al. 1998, Friedman and Auble 2000, Nilsson and Berggren 2000). For example, cottonwood (*Populus* spp.), willow (*Salix* spp.), and many other riparian species native to North America are pioneers that colonize bare sites produced by flood disturbance. By reducing flood magnitude and frequency, dams decrease establishment opportunities for such species (Johnson 1992) and can improve the competitive ability of shade-tolerant exotic species that do not depend upon disturbance, such as Russian-olive (*Elaeagnus angustifolia*; Katz 2001). However, even if dam removal reduces available habitat for seedlings of exotic species, established adults may persist for decades until a flood, drought, age-related factors, or some other agent kills them. Persistence of

large woody plants established under the former regulated flow regime could indefinitely impede the resumption of channel movement after dam removal because of their stabilizing influence on channel banks.

Case study: Elwha River, Washington. Large quantities of sediment are predicted to be transported downstream following the proposed removal of the Elwha Dam and Glines Canyon Dam on the Elwha River, Washington (Hoffman and Winter 1996). Results of current sediment modeling efforts (USDOI 1996) predict that 15% to 35% of the coarse sediment (sand, gravel, and cobbles) and about half of the fine sediment (silt- and clay-size particles) would be eroded from the two reservoirs following dam removal. The remaining sediment would be left along the reservoir margins as a series of terraces. Fine-sediment concentrations released from the reservoirs would be high during periods of dam removal, typically 200 to 1000 parts per million (ppm) but occasionally as high as 30,000 to 50,000 ppm. After the dams are removed, fine sediment concentrations would be low during periods of low flow and high during flood flows that erode channels in the reservoir areas. Within 2 to 5 years, concentrations would return to natural levels. Coarse sediment would aggrade in the relatively steep reaches of the river up to 15 centimeters (cm). Sediment aggradation in moderate-gradient alluvial reaches would promote natural patterns of lateral channel migration, especially near the river's mouth. Over the short term (up to 5 years), this could potentially increase river stages during the 100-year flood up to 1 m. Over the long term (50 years), aggradation could continue and increase existing river stages during the 100-year flood up to 1.5 m with an average increase of 0.75 m. Coarse sediment would enlarge the delta at the river's mouth to a size and character similar to that of predam conditions. As sediment modeling of this basin advances over the years, estimates of the magnitude and timing of sediment transport will become more refined. Yet current results provide an effective framework for predicting vegetation responses to dam removal.

Currently, red alder (*Alnus rubra*) is much more prevalent than black cottonwood (*Populus trichocarpa*) and native willows (*Salix* spp.) along the Elwha River downstream of the dams (figure 3). On the basis of predicted changes in fluvial geomorphology following dam removal, it appears that *Populus* and *Salix* would be favored in the colonization of alluvial reaches of the Elwha River. The life history, ecology, and physiology of these genera are well adapted to the natural flow regimes and sediment-deposition patterns predicted for the Elwha River (Braatne et al. 1996). The relatively high volumes of sediment transport and deposition in alluvial reaches subsequent to dam removal will not favor red alder. Several studies have shown that red alder is vulnerable to hypoxic conditions arising from sediment deposition or extended periods of inundation (Harrington et al. 1994). Therefore, a decrease in red alder and an increase in black cottonwood and willow would be expected in alluvial reaches following dam removal. Additional evidence for these changes in riparian vegetation

can be found in the extensive cottonwood forests of the Dungeness River, an adjacent, undammed basin on the Olympic Peninsula of Washington (Dunlap 1991).

Upstream responses

Vegetation within the former reservoir pool.

Upstream of the dam, dam removal exposes areas of bare ground that were formerly under water, and river discharge (rather than reservoir storage) controls water stages. This will generally produce shifts from the always inundated aquatic zone to mostly inundated and occasionally inundated wetland and riparian vegetation zones, and from inundated or groundwater-affected zones to upland vegetation (figure 1). Thus dam removal may lead to mortality of vegetation along the former reservoir margin, especially if it is sensitive to water table declines associated with the drawdown. The distribution and location of changes in hydroperiods will depend on the topography and stage–discharge relations that develop following dam removal. In many cases, accumulation of sediment behind the reservoir will have altered the topography. If the new stream channel downcuts to near its previous elevation faster than the overall area erodes, then the distribution of hydroperiods in the reservoir pool may be drier following dam removal than before the dam was constructed (Lenhart 2000). On the other hand, partial dam removals in which a lowered control structure is left in place will yield a new storage capacity and effective stage–volume relation and could produce a new set of hydroperiods that may be wetter than those of the predam river.

Initially, vegetation is unlikely to be in equilibrium with the new distribution of hydroperiods. Rather, there will be a transition phase involving colonization of extensive bare areas or mud flats uncovered as water stages decline with the draining of the reservoir (figure 4). Dense, natural revegetation of these areas during the growing season has been observed within weeks in humid regions (AR/FE/TU 1999), while vegetation cover can take years to recover in less productive settings, such as subalpine reservoir margins in the Rocky Mountains (Mansfield 1993). Propagules of early colonizing plants may be present in seed banks or may be dispersed from adjacent areas. The initial colonizing plants can have a substantial long-term influence on plant composition through the persistence of long-lived individuals, vegetative reproduction, relatively higher seed production of those



Figure 3. Young red alder trees (*Alnus rubra*) line the channel and midchannel bars of the Elwha River, Washington, while older black cottonwoods (*Populus trichocarpa*) occupy older, higher surfaces. Conditions resulting from proposed dam removals on the Elwha River could lead to a decrease in red alder and an increase in black cottonwood. Photograph by Patrick Shafroth.

species, and alterations of the physical environment (Mansfield 1993). Initial plant colonists of sites characteristic of former reservoir bottoms (bare, moist, nutrient-rich, with a depauperate seed bank) tend to be weedy plants with typical ruderal traits such as rapid growth, high levels of seed production, and effective dispersal mechanisms. This group of plants may include a relatively high fraction of invasive, non-native species (Galatowitsch et al. 1999, Lenhart 2000).

Case study: Removal of small dams in Wisconsin.

Many small dams in the northeast and upper Midwest were built between the mid-1800s and early 1900s to power lumber and flour mills. Because of abundant water resources and the early development of dams for mechanical and small-scale hydroelectric energy, the state of Wisconsin has more than 3600 dams. Safety and economic reasons (i.e., where repair costs greatly exceeded removal costs) have led to the removal of more than 70 dams since 1950 in Wisconsin (Born et al. 1998, AR/FE/TU 1999).

Lenhart (2000) performed a retrospective analysis of natural vegetation recolonization in five former impoundments in Wisconsin. Two sites represented long-term (more than 40 years) recovery periods, whereas three sites had recovered in 3 to 5 years. Across all sites, high-nutrient sediments, ranging in depth from 25 to 200 cm, had been deposited over predam soils. Vegetation at the three younger sites had low species diversity and were dominated by large, monotypic stands of pioneer species like stinging nettle (*Urtica dioica*), reed canary grass (*Phalaris arundinacea*), and rice-cut grass (*Leersia oryzoides*). The plant communities observed on the

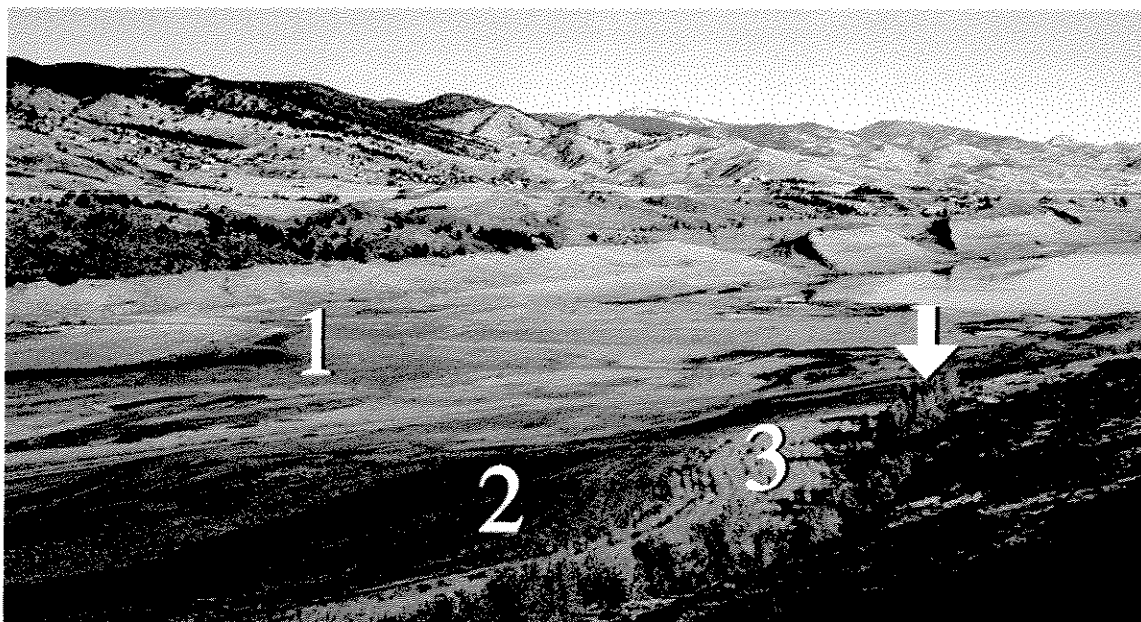


Figure 4. Vegetation colonization on the exposed bottom of Horsetooth Reservoir, Colorado. Between January 2000 and October 2001, water was drawn down 32 meters to enable dam repairs, reducing the water surface area from 621 to 77 hectares. Numbers refer to bands of vegetation dominated by the following non-native species: (1) goosefoot (*Chenopodium glaucum*), (2) smartweed (*Polygonum lapathifolium* and *P. persicaria*), (3) sweet clover (*Melilotus* spp.). The arrow points to mature cottonwood trees (*Populus deltoides*) that approximate the high water line. Photograph by Patrick Shafroth.

younger sites did not resemble any native communities. Young sites tended to be composed of a high fraction of wetland plants, which colonized the moist surfaces that were exposed following dam removal. Over time, sites became drier and were dominated by more xeric species. The two older sites had higher species diversity but included a higher percentage of nonnative species.

Management considerations

Restoration potential. Dam removal should not always be expected to restore riparian ecosystems to their predam condition (figure 5). A spectrum of outcomes is possible, given the variability in predam conditions, the responses of the system to the dam, and the responses to dam removal (Zedler 1999). Ecological systems frequently exhibit hysteresis and time-lagged responses, the details of which are not clear with respect to riparian vegetation, although a transient phase of 50 to 100 years has been observed when systems respond to dam construction and operation (Petts 1987, Johnson 1998). Legacies of flow regulation such as altered channel morphology, species composition, and age structure may result in a delayed response of the system to naturalized flows. Even if dam removal restored the natural flow regime, effects of dam removal would vary regionally with factors such as climate, flood regime, geology, and fluvial processes associated with riparian vegetation establishment (Friedman and Auble 2000). Other anthropogenic impacts to a river system, such as adjacent groundwater pumping, channel stabilization, and agri-

cultural and residential development, could prevent a return to predam conditions (figure 5). Effects of extreme events that occurred before but not during the dammed period (Katz 2001) or climate differences in the predam and postdam removal periods could also influence the response. Despite these possible limitations, dam removal has the potential to restore valuable components of riparian ecosystems, and some management actions could enhance this potential.

Managing for a beneficial transient sediment pulse. In some dam removal situations, relatively small pulses of sediment could promote enough channel change to create surfaces suitable for the establishment of riparian forest, without greatly damaging other resources. It could be argued that there is little value in managing for a transient benefit, because eventually trees established as a result of the sediment pulse would die. However, this view underestimates the importance of transient events in structuring populations of disturbance-dependent, long-lived species. For example, the cottonwood gallery forests along the Platte River system are a product of an adjustment in channel size following water management (Johnson 1998). Establishment of these forests was a transient event, not an equilibrium expression of the predam or postdam flow or sediment regime. Once established, such forests exist for more than a century, which is longer than the life of many dams. Given the persistent effects of transient events in these ecosystems, managing the sediment pulse following dam removal could be an efficient conservation strategy.

Controlling the reservoir

drawdown. The timing and pattern of drawdown heavily influences the species composition of bare, moist areas by exposing sites at times that do or do not match the life history characteristics of various species with respect to germination and early seedling establishment requirements. Much practical experience with manipulating drawdowns to achieve desired mixes of herbaceous species is embodied in the wildlife-management strategy of "moist soil management" (Fredrickson and Taylor 1982). Many refuges and waterfowl management areas actively manipulate drawdowns in shallow constructed impoundments or moist soil units to grow specific species with desired food and cover value for wildlife. Similar approaches have been effectively employed in riparian restoration efforts to encourage natural establishment of desired native trees and shrubs (Roelle and Gladwin 1999). In arid and semiarid landscapes, where seedling establishment requirements for native riparian trees are often

much wetter than the conditions they require as adults, the plants established during the transition or drawdown phase may persist and dominate the drier postdam regime for many decades. Recruitment of cypress (*Taxodium distichum*) and tupelo (*Nyssa aquatica*), after extended drawdown of a large impoundment in the southeastern United States suggests that natural establishment of bottomland hardwood forest could be expected following dam removal, assuming there are upstream sources of seed, that large numbers of seeds were produced the previous season, and that subsequent water levels do not exceed average seedling height for extended periods (Keeland and Conner 1999). Few dam removal projects have attempted to manipulate the timing and pattern of drawdown during the transition phase so as to produce desired vegetation. Where the reservoir pool can be lowered by draining and pumping before any work is done on the dam structure, there is tremendous potential for effective, even multiyear control over the plant community by managing water stages during the transition phase (ASCE 1997).

Invasive species. Although dam removals represent a significant opportunity for riparian habitat restoration, they also provide opportunities for invasion of undesirable, nonnative species (figure 4; Galatowitsch et al. 1999, Lenhart 2000). High levels of physical disturbance result in significant proportions of exotic species in many riparian floras (Planty-

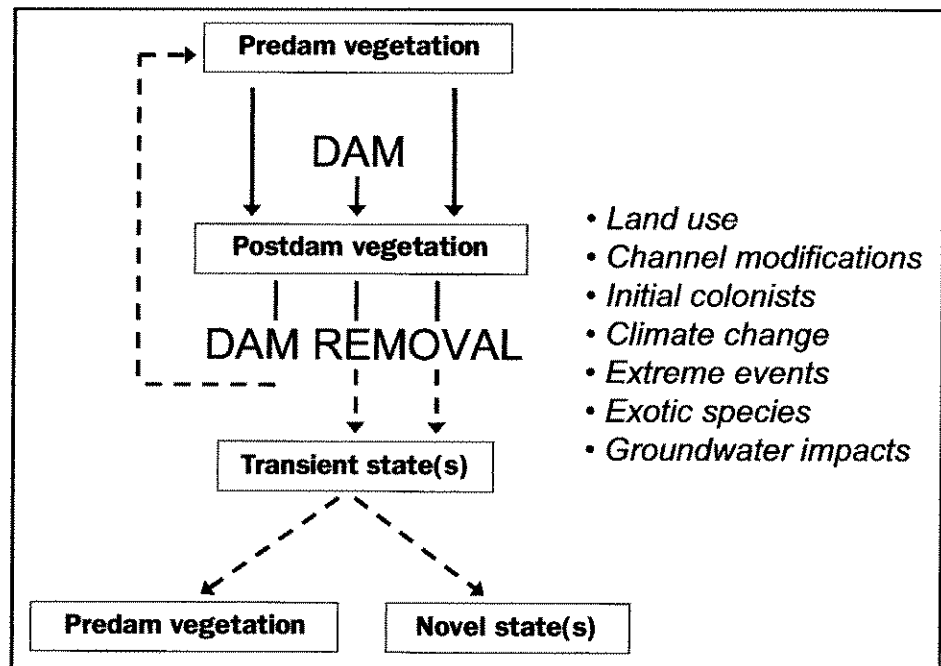


Figure 5. Multiple pathways of riparian vegetation change from unregulated conditions through postdam removal states. Riparian vegetation may respond to dam construction and operation in various ways, and multiple trajectories are possible following dam removal, depending on initial conditions and the nature of hydrologic and geomorphic change. Other factors, including those listed next to the flow diagram, also influence riparian vegetation response. As a result, in many cases, riparian vegetation is unlikely to quickly return to its predam condition.

Tabacchi et al. 1996, Tickner et al. 2001). The extensive, bare, nutrient-rich sediments of the former impoundment provide a substrate that may favor weedy, nonnative plants (Dukes and Mooney 1999). Once established, nonnative weeds may inhibit the establishment of native species, thus reducing plant and animal species diversity (Galatowitsch et al. 1999, Middleton 1999) and influencing succession (Hobbs and Mooney 1993). Where the risk of nonnative vegetation establishment is high, a more managed approach to vegetation establishment following dam removal may be warranted.

Active revegetation. Dam removal plans may include broadcast seeding or limited tree planting aimed at precluding the establishment of undesirable nonnative species or stabilizing sediments in the former reservoir pool (ASCE 1997, AR/FE/TU 1999). Additional reasons for active revegetation following dam removal include creating habitat diversity and improving recreational use. Secondary measures such as installation of structures to slow or reduce bank erosion, construction of fenced enclosures to manage livestock, and multiyear irrigation of plantings have been necessary elements of revegetation efforts in arid and semiarid regions of the United States (Briggs 1996). Active revegetation of riparian shrubs and trees in the western United States has often failed because of insufficient understanding of establishment and survival requirements of native species and continued live-

stock grazing following planting (Kauffman et al. 1995, Briggs 1996).

Plantings of early successional native species with relatively high growth rates may be an effective means of minimizing the establishment of exotic plant species and initiating natural successional processes. Dense stands of native woody plants, such as cottonwood and willow, may effectively shade out and thus exclude many exotic herbaceous annual and perennial plants. In contrast, planting slow growing, late-successional or climax species following dam removal may provide exotic weeds with an initial advantage. In the mid-western United States, plants such as smartweeds (*Polygonum* spp.), rice-cut grass, barnyard grass (*Echinochloa crus-galli*), and sod-forming sedges (*Carex* spp.) often naturally recolonize disturbed prairie wetlands. Other species, which may effectively compete with aggressive weeds, have been suggested for planting as potential native cover crops. These include late-season grasses such as *Spartina pectinata* and forbs such as *Coreopsis* spp. and *Ratibida* spp. (Galatowitsch and van der Valk 1994). Cover crops may quickly occupy sites, stabilizing the soil surface and usurping space that might otherwise be taken by less desirable species. In subsequent years, more slowly growing species may gradually replace the annuals. In the southwestern United States, attempts to actively restore native riparian understory species by planting, removal of non-natives, and use of commercial soil amendments were ineffective, largely because of the rapid regrowth or establishment of nonnative species already on site (Wolden and Stromberg 1997). Recommendations for future efforts suggested that (a) seeding should be done over several years to accommodate climatic and hydrologic variability, (b) seed mixes should include species reflecting a diversity of life-history traits so species can sort out across the range of fine-scale environmental conditions that may exist at the restoration site, and (c) some weedy native annuals may compete well initially with non-natives.

The assumption that a diverse set of species will naturally disperse to and become established on a site following the planting of a few of the dominant species is not always valid—such planting has produced stands of relatively low diversity in reforested bottomland hardwood forests (Allen 1997). Experimentation can make seed selection more efficient by helping to determine which species will recruit well naturally versus which need to be planted and which and how many species are necessary to develop desired ecosystem functions (Zedler et al. 2001).

Ultimately, a fundamental goal of any attempt to actively reestablish self-sustaining wetland and riparian vegetation should be to restore or reestablish key physical processes such as natural flow variability and channel change (Middleton 1999, Stromberg 2001). Such physical processes integrate terrestrial and aquatic elements of the watershed, producing spatially and temporally distinctive patterns of vegetation establishment (Scott et al. 1996). Restoration of key physical processes, in concert with active revegetation, enhances long-term success. The displacement of native wetland and ripar-

ian vegetation by invasive, nonnative species is typically associated with alteration of the natural hydrologic regime and land use practices that reduce flooding, lower water tables, and alter soil properties (Briggs 1996). Efforts aimed at actively revegetating herbaceous (Wolden and Stromberg 1997, Middleton 1999) and woody (Briggs 1996) vegetation have benefited from natural flooding.

Research needs

There is a strong need for more quantitative studies of the response of vegetation to dam removal. This should include rigorous monitoring of new or recent dam removals or retrospective analyses of older sites. Long-term studies will be necessary to elucidate potentially complex pathways of vegetation change. The potential for the generation of novel plant communities associated with the unusual physical conditions that may follow dam removal represents an intriguing topic of ecological research. Manipulative experiments could be used to test different management techniques, including controlled drawdowns and various planting approaches. Given the well-documented importance of fluvial geomorphic and hydrologic conditions in structuring riparian vegetation, botanists and plant ecologists should seek collaborations with physical scientists and couple plant response models to models used to estimate water and sediment dynamics following dam removal.

Acknowledgments

Thanks to David Hart and LeRoy Poff for organizing the dam removal symposium at the 2001 annual meeting of the Ecological Society of America and to The Pew Charitable Trusts for providing support that helped us to complete this article. Thanks also to those who shared their experiences with dam removal, including Madeline Fisher, Jeff Connor, and Chris Lenhart. The comments of David Hart and three anonymous reviewers improved the manuscript. This manuscript was prepared by employees of the US Department of the Interior, US Geological Survey, and therefore may not be copyrighted.

References cited

- Allen JA. 1997. Reforestation of bottomland hardwoods and the issue of woody species diversity. *Restoration Ecology* 5: 125–134.
- [AR/FE/TU] American Rivers, Friends of the Earth, Trout Unlimited. 1999. *Dam Removal Success Stories: Restoring Rivers through Selective Removal of Dams That Don't Make Sense*. Washington (DC): AR/FE/TU.
- [ASCE] American Society of Civil Engineers. 1997. *Guidelines for retirement of dams and hydroelectric facilities*. New York: ASCE.
- Barrick KA, Noble MG. 1993. Micronutrient status of tree species affected by the Lawn Lake flood in Rocky Mountain National Park, Colorado. Pages 70–85 in McCutchen HE, Herrmann R, Stevens DR, eds. *Ecological Effects of the Lawn Lake Flood of 1982, Rocky Mountain National Park*. Washington (DC): US Department of the Interior, National Park Service. Scientific Monograph NPS/NRRMO/NRSM-93/21.
- Bednarek AT. 2001. Undamming rivers: A review of the ecological impacts of dam removal. *Environmental Management* 27: 803–814.
- Born SM, Genskow KD, Filbert TL, Hernandez-Mora N, Keefer MI, White KA. 1998. Socioeconomic and institutional dimensions of dam removals: The Wisconsin experience. *Environmental Management* 22: 359–370.

- Braatne JH, Rood SB, Heilman PE. 1996. Life history, ecology, and conservation of riparian cottonwoods in North America. Pages 423–458 in Stetler RF, Bradshaw HD Jr, Heilman PE, Hindley TM, eds. *Biology of Populus and Its Implications for Management and Conservation*. Ottawa (Canada): National Research Council Research Press.
- Briggs MK. 1996. *Riparian Ecosystem Recovery in Arid Lands: Strategies and References*. Tucson: University of Arizona Press.
- Dukes JS, Mooney HA. 1999. Does global change increase the success of biological invaders? *Trends in Ecology and Evolution* 14: 135–139.
- Dunlap JM. 1991. Genetic variation in natural populations of *Populus trichocarpa* from four river valleys (Hoh, Dungeness, Nisqually and Yakima) in Washington State. PhD dissertation. University of Washington, Seattle, WA.
- Fredrickson LH, Taylor TS. 1982. Management of Seasonally Flooded Impoundments for Wildlife. Washington (DC): US Fish and Wildlife Service. Resource Publication no. 148.
- Friedman JM, Auble GT. 2000. Floods, flood control, and bottomland vegetation. Pages 219–237 in Wohl E, ed. *Inland Flood Hazards: Human, Riparian and Aquatic Communities*. Cambridge (UK): Cambridge University Press.
- Galatowitsch SM, van der Valk AG. 1994. *Restoring Prairie Wetlands: An Ecological Approach*. Ames: Iowa State University Press.
- Galatowitsch S, Budelsky R, Yetka L. 1999. Revegetation strategies for north temperate glacial marshes and meadows. Pages 225–241 in Streever W, ed. *An International Perspective on Wetland Rehabilitation*. Dordrecht (The Netherlands): Kluwer Academic Publishers.
- Gilbert GK. 1917. Hydraulic Mining Debris in the Sierra Nevada. Washington (DC): US Geological Survey. Professional Paper no. 105.
- Halpern CB, Harmon ME. 1983. Early plant succession on the Muddy River mudflow, Mount St. Helens, Washington. *American Midland Naturalist* 110: 97–106.
- Harrington CA, Zasada JC, Allen EA. 1994. Biology of red alder (*Alnus rubra* Bong.). Pages 3–22 in Hibbs DE, DeBell DS, Tarrant RF, eds. *The Biology and Management of Red Alder*. Corvallis: Oregon State University Press.
- Hobbs RJ, Mooney HA. 1993. Restoration ecology and invasions. Pages 127–133 in Saunders DA, Hobbs RJ, Ehrlich PR, eds. *Nature Conservation 3: Reconstruction of Fragmented Ecosystems*. Chipping Norton (Australia): Surrey Beatty and Sons.
- Hoffman CH, Winter BD. 1996. Restoring aquatic environments: A case study of the Elwha River. Pages 303–323 in Wright RG, ed. *National Parks and Protected Areas: Their Role in Environmental Protection*. Cambridge (MA): Blackwell Scientific.
- Hotchkiss RH, Barber ME, Wohl EE. 2001. Dam decommissioning: Decisions and unresolved sediment transport issues. Pages IX-64–IX-71 in *Proceedings of the Seventh Federal Interagency Sedimentation Conference*; 25–29 March 2001; Reno, NV.
- Hupp CR. 1988. Plant ecological aspects of flood geomorphology and paleoflood history. Pages 335–356 in Baker VR, Kochel C, Patton PC, eds. *Flood Geomorphology*. New York: Wiley Interscience.
- James LA. 1989. Sustained storage and transport of hydraulic gold mining sediment in the Bear River, California. *Annals of the Association of American Geographers* 79: 570–592.
- Jansson R, Nilsson C, Dynesius M, Anderson E. 2000. Effects of river regulation on river-margin vegetation: A comparison of eight boreal rivers. *Ecological Applications* 10: 203–224.
- Jarrett RD, Costa JE. 1993. Hydrology and geomorphology of the 1982 Lawn Lake dam failure, Colorado. Pages 1–17 in McCutchen HE, Herrmann R, Stevens DR, eds. *Ecological Effects of the Lawn Lake Flood of 1982, Rocky Mountain National Park*. Washington (DC): US Department of the Interior, National Park Service. Scientific Monograph NPS/NRROMO/NRSM-93/21.
- Johnson WC. 1992. Dams and riparian forests: Case study from the upper Missouri River. *Rivers* 3: 229–242.
- . 1998. Adjustment of riparian vegetation to river regulation in the Great Plains, USA. *Wetlands* 18: 608–618.
- Kalliola R, Salo J, Puhakka M, Rajasilta M. 1991. New site formation and colonizing vegetation in primary succession on the western Amazon floodplains. *Journal of Ecology* 79: 877–901.
- Katz GL. 2001. Fluvial disturbance, flood control, and biological invasion in Great Plains riparian forests. PhD dissertation. University of Colorado, Boulder, CO.
- Kauffman JB, Case RL, Lytjen D, Otting N, Cummings DL. 1995. Ecological approaches to riparian restoration in northeast Oregon. *Restoration and Management Notes* 13: 12–15.
- Keddy PA, Reznicek AA. 1986. Great Lakes vegetation dynamics: The role of fluctuating water level and buried seeds. *Journal of Great Lakes Resources* 12: 25–36.
- Keeland BD, Conner WH. 1999. Natural regeneration and growth of *Taxodium distichum* (L.) Rich. in Lake Chicot, Louisiana after 44 years of flooding. *Wetlands* 19: 149–155.
- Keigley RB. 1993. Observations of plant ecology on the Lawn Lake flood alluvial fan. Pages 192–214 in McCutchen HE, Herrmann R, Stevens DR, eds. *Ecological Effects of the Lawn Lake Flood of 1982, Rocky Mountain National Park*. Washington (DC): US Department of the Interior, National Park Service. Scientific Monograph NPS/NRROMO/NRSM-93/21.
- Krasny ME, Vogt KA, Zasada JC. 1988. Establishment of four Salicaceae species on river bars in interior Alaska. *Holarctic Ecology* 11: 210–219.
- Leck MA. 1989. Wetland seed banks. Pages 283–305 in Leck MA, Parker VT, Simpson RL, eds. *Ecology of Soil Seed Banks*. San Diego (CA): Academic Press.
- Lenhart CF. 2000. The vegetation and hydrology of impoundments after dam removal in southern Wisconsin. Master's thesis. University of Wisconsin, Madison, WI.
- Madej MA, Ozaki V. 1996. Channel response to sediment wave propagation and movement, Redwood Creek, California, USA. *Earth Surface Processes and Landforms* 21: 911–927.
- Major JJ, Pierson TC, Dinehart RL, Costa JE. 2000. Sediment yield following severe volcanic disturbance; a two-decade perspective from Mount St. Helens. *Geology* 28: 819–822.
- Mansfield DH. 1993. Vegetation development on the exposed shores of Lawn Lake. Pages 86–111 in McCutchen HE, Herrmann R, Stevens DR, eds. *Ecological Effects of the Lawn Lake Flood of 1982, Rocky Mountain National Park*. Washington (DC): US Department of the Interior, National Park Service. Scientific Monograph NPS/NRROMO/NRSM-93/21.
- Marschner H. 1995. *Mineral Nutrition of Higher Plants*. San Diego (CA): Academic Press.
- Middleton B. 1999. *Wetland Restoration: Flood Pulsing and Disturbance Dynamics*. New York: John Wiley and Sons.
- Naiman RJ, Decamps H. 1997. The ecology of interfaces: Riparian zones. *Annual Review of Ecology and Systematics* 28: 621–658.
- Nilsson C, Berggren K. 2000. Alterations of riparian ecosystems caused by river regulation. *BioScience* 50: 783–792.
- Petts GE. 1987. Time-scales for ecological change in regulated rivers. Pages 257–266 in Craig JE, Kemper JB, eds. *Regulated Streams: Advances in Ecology*. New York: Plenum Press.
- Pizzuto J. 2002. Effects of dam removal on river form and process. *BioScience* 52: 683–691.
- Planty-Tabacchi A-M, Tabacchi E, Naiman RJ, DeFerrari C, Decamps H. 1996. Invasibility of species-rich communities in riparian zones. *Conservation Biology* 10: 598–607.
- Poff NL, Allan JD, Bain MB, Karr JR, Prestegard KL, Richter BD, Sparks RE, Stromberg JC. 1997. The natural flow regime: A paradigm for river conservation and restoration. *BioScience* 47: 769–784.
- Roelle JE, Gladwin DN. 1999. Establishment of woody riparian species from natural seedfall at a former gravel pit. *Restoration Ecology* 7: 183–192.
- Rood SB, Kalischuk AR, Mahoney JM. 1998. Initial cottonwood seedling recruitment following the flood of the century of the Oldman River, Alberta, Canada. *Wetlands* 18: 557–570.
- Scott ML, Friedman JM, Auble GT. 1996. Fluvial process and the establishment of bottomland trees. *Geomorphology* 14: 327–339.

- Semmens DJ, Osterkamp WR. 2001. Dam removal and reservoir erosion modeling: Zion Reservoir, Little Colorado River, AZ. Pages IX-72-IX-79 in Proceedings of the Seventh Federal Interagency Sedimentation Conference; 25-29 March 2001; Reno, NV.
- Simons RK, Simons DB. 1991. Sediment problems associated with dam removal—Muskegon River, Michigan. Pages 680-685 in Hydraulic Engineering: Proceedings of the 1991 National Conference of the American Society of Civil Engineers. New York: American Society of Civil Engineers.
- Stromberg JC. 2001. Restoration of riparian vegetation in the southwestern United States: Importance of flow regimes and fluvial dynamism. *Journal of Arid Environments* 49: 17-34.
- Taylor JP, Wester DB, Smith LM. 1999. Soil disturbance, flood management, and riparian woody plant establishment in the Rio Grande floodplain. *Wetlands* 19: 372-382.
- Tickner DP, Angold PG, Gurnell AM, Mountford JO. 2001. Riparian plant invasions: Hydrogeomorphological control and ecological impacts. *Progress in Physical Geography* 25: 22-52.
- [USDOI] US Department of the Interior. 1996. Sediment Analysis and Modeling of the River Erosion Alternative. Washington (DC): US Bureau of Reclamation. Elwha Technical Series no. PN-95-9.
- Walker LR, Zasada JC, Chapin FS. 1986. The role of life history processes in primary succession on an Alaskan floodplain. *Ecology* 67: 1243-1253.
- Williams GP, Wolman MG. 1984. Downstream Effects of Dams on Alluvial Rivers. Washington (DC): US Geological Survey. Professional Paper 1286.
- Wohl EE, Cenderelli DA. 2000. Sediment deposition and transport patterns following a reservoir sediment release. *Water Resources Research* 36: 319-333.
- Wolden LG, Stromberg JC. 1997. Experimental treatments (and unplanned natural events) for restoration of the herbaceous understory in an arid-region riparian ecosystem. *Restoration and Management Notes* 15: 161-167.
- Zedler JB. 1999. The ecological restoration spectrum. Pages 301-318 in Streever W, ed. *An International Perspective on Wetland Rehabilitation*. Dordrecht (The Netherlands): Kluwer Academic.
- Zedler JB, Callaway JC, Sullivan G. 2001. Declining biodiversity: Why species matter and how their functions might be restored in Californian tidal marshes. *BioScience* 51: 1005-1017.

Enlisting the Social Sciences in Decisions about Dam Removal

SARA E. JOHNSON AND BRIAN E. GRABER

Ecologists and conservationists share the desire to have healthy and sustainable ecosystems. But many of our society's ecological management activities and policies thus far have not resulted in sustainable ecosystems, and ecologists and conservationists recognize that considerable change in human behavior will be required to achieve that goal. It would stand to reason, then, that ecologists and conservationists would also share a desire for decisionmaking that considers alternatives, such as selectively removing dams, that could result in outcomes beneficial to ecosystems.

Experience, however, shows that decisions on ecological issues often do not include alternatives that could benefit ecosystem health. When they do, such alternatives are often dead on arrival and are not given serious consideration or adequate review. Experiences with local decisions concerning old and obsolete small dams (for discussion purposes, dams 7.6 meters high or less) highlight this problem. Unlike decisions regarding larger dams, which are typically made in a court of law and are based on expert testimony, decisions affecting the future of small dams are usually made in a local "court of public opinion" and involve many stakeholders and decision-makers, including private citizens and citizen groups, elected officials, government resource agency personnel, and local business interests. While certain individuals or bodies, such as the private dam owner, agency personnel, or elected officials, may have actual legal authority to make the final decision, public support, or lack of it, can make or break a restoration opportunity.

Research shows that this kind of decisionmaking about the future of dams and rivers is often poor. Born and colleagues (1998) looked at 14 dam removal cases in Wisconsin and found that decisions were commonly made with incomplete and inaccurate information and in emotionally charged and divisive atmospheres. These findings support our experiences that the divisiveness of decisionmaking is exacerbated

THE APPLICATION OF SOCIAL SCIENCE CONCEPTS AND PRINCIPLES TO PUBLIC DECISIONMAKING ABOUT WHETHER TO KEEP OR REMOVE DAMS MAY HELP ACHIEVE OUTCOMES LEADING TO SUSTAINABLE ECOSYSTEMS AND OTHER GOALS IN THE PUBLIC INTEREST

when one or more of the following situations exists: when the idea of removal is new to the community; when the dam poses public safety concerns, thus forcing a quick decision; and when outsiders (e.g., state agency personnel or conservation organizations with representatives not from the area) are involved in the decision process.

Faced with the uncertainty that such circumstances are likely to produce and the need to make decisions, humans commonly resort to psychological shortcuts to help make those decisions (Cialdini 2001). Such shortcuts include accepting the prevailing social norm as one's own position, adopting the opinion of someone who is similar to oneself and

Sara E. Johnson (e-mail: johnson@tu.org) is director of volunteer operations for Trout Unlimited, 1500 Wilson Boulevard, Arlington, VA 22209-2404, and cofounder and former executive director of the River Alliance of Wisconsin. Brian E. Graber (e-mail: bgrab@tu.org) is a consulting watershed restoration specialist with a background in fluvial geomorphology and hydrology; his address is 211 S. Paterson Street, Madison, WI 53703. © 2002 American Institute of Biological Sciences.

is well liked, or simply digging in one's heels to remain consistent with one's earlier actions or words. These and other psychological principles direct human behavior, generally without an awareness that they are being used.

We are not suggesting that dam removal is always the best alternative; indeed, in some cases, removal of a dam could cause long-term harm to the ecosystem. But because dams can have a profound and often negative impact on water quality, river function, and ecology (Baxter 1977, Graf 1980, Petts 1980, Petts and Pratts 1983, Williams and Wolman 1984, Chien 1985, Andrews 1986, Ligon et al. 1995, Power et al. 1996, Hadley and Emmett 1998, Brandt 2000, Graf 2001, Magilligan and Nislow 2001), decisionmaking about dam removals should be improved. In an improved process, alternatives (such as selective dam removal, where it might benefit the ecosystem) should be considered and informed by scientific findings about potential outcomes (social, economic, and environmental), so that alternatives may be accepted or rejected on their merits (AR/TU 2002).

Social science principles and practices have long been used to encourage societal change in the areas of public health and safety, but their use to encourage beneficial change in the health of the ecosystem has been slow to take hold. However, if fundamental social and psychological principles such as these decision shortcuts are understood and factored into the design of public information efforts and decisionmaking, they may hold substantial potential to influence social change concerning dams and rivers, such that "win-win-win" outcomes may be achieved that benefit not only the ecosystem but also dam owners and the local community.

We look briefly at the changing socioeconomic context in which public decisions around dams and rivers are being made, explore the use of social science concepts and principles to improve such decisions, and discuss potential roles of scientists in public decisionmaking that affects the sustainability of ecosystems. The use of concepts and principles discussed here is not limited to decisions about dams and rivers; they could be helpful in any decisionmaking that could lead to improved ecosystem health or other public benefits.

What's the big deal about small dam removal?

A number of social, economic, and environmental factors are converging in a manner that is raising public awareness of dams and their impacts on rivers and streams. A brief discussion of some of these factors provides a societal context for decisions about dams and rivers.

Societal values (and associated economic values) regarding dams and rivers have changed over time. Changing societal needs and technological changes over the past century, for example, have left many dams, especially smaller structures, no longer useful for their original purpose and in need of extensive and potentially costly repairs. Estimates of the total number of dams in the United States range from 76,000 (NID 2001) to 2.5 million (Johnston 1992); one reason for this disparity is that many smaller dams are not included in na-

tional databases or even in some state databases. The American Society of Civil Engineers recently rated the safety of America's dam infrastructure with a grade of "D," citing 61 reported failures over the past two years (ASCE 2001). Many smaller dams, built over 100 years ago and no longer used for their original purpose, such as grist milling or raising water levels to float logs to timber mills, have deteriorated to the point that they pose public safety hazards, which has led states to order repairs or removal to alleviate the hazard.

Dams have a finite life expectancy, often stated to be on the order of 50 years (FEMA 1999, ASDSO 2001), and even many of those built more recently than the old grist mills have reached or are approaching the end of their useful lives, although repairs can maintain a structure for longer periods. But costs to repair or rebuild a deteriorating small dam are typically high—from hundreds of thousands to even millions of dollars in some cases (Born et al. 1998, AR/FE/TU 1999, TU 2001). The high cost of maintaining old dams, especially obsolete ones, is forcing dam owners (often municipalities) to look for solutions.

As small dams have become less critical to the US infrastructure and the financial costs of maintaining them have become greater, society's understanding and appreciation of the values of healthy waters in general have grown. The growth of water-based outdoor recreation, for example, coincided with water quality improvements after passage of the Clean Water Act in 1972; annually, more than 35 million people fish (Fedler 2000) and 25 million canoe or kayak (ACA 2000). Scientists in recent decades have enabled greater understanding of the vital role of naturally functioning river systems in the context of ecosystem health and sustainability; resource agencies have even reorganized by watershed boundaries rather than by political boundaries.

Experience and some research show that selective removal of small dams is one method for river ecosystem restoration that can be in the public interest. Documented public benefits of selective dam removal include cost-effective water quality improvements; cost-effective and permanent removal of a public safety hazard; cost-effective restoration of fish and wildlife habitat for endangered species or sport fisheries (or both); recreational improvements; aesthetic enhancement, such as restoration of waterfalls or riffles; and opportunities for community economic revitalization and associated quality-of-life enhancements (AR/FE/TU 1999, TU 2001).

It is within this changing socioeconomic context that more and more local communities are facing decisions concerning their dams. The difficulty of deciding may in part be a reflection of these changing "bigger picture" factors running up against local communities with strong attachments to their dam and its impoundment and a strong preference for the status quo. In some cases, removal of the dam, even an obsolete structure, is not even considered to be an option.

More and more small dams are being removed nonetheless, primarily to relieve the economic burden of deteriorating structures and to eliminate public safety hazards, but

also to meet concerns about the environment and conservation, especially to improve water quality or restore native or sport fisheries (AR/FE/TU 1999). Many deliberations about small dam repair or removal still result in a decision to repair old and sometimes obsolete structures. In some cases, economic, historical, environmental, or other factors may warrant repairing a dam. But in many cases, such decisions to keep the structure are made at great expense to the river when the water quality and fisheries continue to degrade; to the dam owners, who are often the taxpayers of the local community; and to local businesses, which might have capitalized on opportunities for economic revitalization with a restored river, especially in urban or downtown settings. Furthermore, the opportunity for restoration of that stretch of water and habitat in a larger ecosystem context may be lost for decades, perhaps even a lifetime.

The issue of dams continues to be pushed higher on the public agenda. In recent years, some elected officials have been attempting to bring dam-related laws into the 21st century; several states (e.g., Pennsylvania, Wisconsin, California) have changed or have attempted to change legislation and policies governing dams and their host rivers in recent years. With dams increasingly in the news, every dam removed for purposes of river system restoration has the potential to make the next one easier or more difficult.

Removing dams is not a new idea; more than 500 dams have been removed in the last century (AR/FE/TU 1999, AR 2001). Nonetheless, it is a new idea to many community members who face a decision about the future of their dam. If river ecosystem restoration is a goal, and selective dam removal is a potential method to achieve that goal, experience suggests that an effective strategy would be to first increase public support for dam removal as a viable tool for river restoration—in short, to influence social change concerning dams and rivers. An efficient approach to social change is necessary to reduce strife in local communities, to avoid unnecessary and expensive financial obligations to dam owners and taxpayers, and to reduce the number of lost opportunities for ecosystem restoration.

Drawing on the social sciences to effect changes in human behavior

The social science literature recognizes important differences between activities that result in increased awareness and understanding and those that result in behavior change. Most public information efforts that are designed to foster change inform to some extent, but seldom are they successful in effecting desired behavior change (Andreasen 1995, Rogers 1995, McKenzie-Mohr and Smith 1999). Not surprisingly, programs thoughtfully and strategically designed to achieve behavior change are more likely to result in actual behavior change.

Following is an overview of some social psychology principles and practices that could be pertinent to efforts to effect changes in human behavior regarding sustainable ecosystems, especially relative to dams and rivers. We look at how

people tend to make decisions when uncertainty is high, how new ideas often spread through communities, and techniques for encouraging acceptance of new ideas at the individual and community levels.

Shortcuts for decisionmaking. Psychologists have long known that when people are asked to make a decision but do not have the desire or the ability to analyze information carefully, they are likely to fall back on psychological “shortcuts.” When these shortcuts are used, the decision to comply or not comply with a request is made on the basis of a single piece of information, such as agreeing if they know their friends or colleagues agree. Cialdini (2001) identifies a number of such triggers for compliance with a request, a few of which we think are especially pertinent to decisions regarding dams and rivers. As is the case with stereotypes, over time an individual judges decision shortcuts to be timesaving and reliable, and he or she is usually unaware of using them.

Social norms as a psychological shortcut. Although researchers have varying definitions of social norms, the term generally refers to what is most often done or approved of in society at large or in a particular setting, such as a local community. In short, people tend to do what others like them are doing. Cialdini (2001) notes that people especially look to see what others are doing when two factors are present: when uncertainty is high and when others exhibiting the behavior are similar to oneself or well liked.

Few programs designed to foster sustainable behavior have taken into consideration the powerful effects of social norms (McKenzie-Mohr and Smith 1999). In many small communities, current social norms concerning dams and rivers appear to support the status quo (that is, keeping the dam) and often preclude the consideration of removal as an alternative. If social norms could be changed to be supportive of healthy and naturally functioning river systems, the alternative of selective dam removal would more likely be considered and then accepted or rejected on its merits.

A practical way to encourage the acceptance of social norms that support dam removal as an alternative, for example, would be to pay close attention to “messengers.” If the aim is to have the support of local business leaders in community A, one approach would be to invite a business leader from a similar community where a river was successfully restored through selective dam removal (community B) to speak to the business leaders of community A. Better yet would be to arrange for community A business leaders to take a field trip to community B to talk with leaders and see the restoration for themselves. This alone would be unlikely to change social norms among the business leaders of community A, but it could be a key component of a successful public information program.

Commitment and consistency as a psychological shortcut. Securing a commitment from someone to do something can be a powerful technique for behavioral change. Written commitment is a stronger motivator than oral commitment, but both can and have resulted in desired behavior

change (Katzev and Wang 1994, McKenzie-Mohr and Smith 1999, Cialdini 2001).

Consistency is closely tied to commitment. Psychologists have recognized for more than 50 years that the desire to be (or to be seen as) consistent is a central motivator of behavior and, therefore, is also a potent component of efforts to influence behavior change (Cialdini 2001).

With the so-called foot-in-the-door technique (Freedman and Fraser 1966) commonly used in sales, commitment to comply with an initial small request leads to a greater likelihood of compliance with a larger request. This marketing technique has also proven effective in energy conservation efforts: Citizens who complied with a request to complete a written survey had a higher rate of compliance when later asked to reduce energy consumption at home (Katzev and Wang 1994). One reason for this greater compliance is that once a person has committed to do something, his or her support for that activity is internalized and therefore becomes even stronger (Cialdini 2001).

Currently, commitment and consistency principles appear to be working against considering dam removal as an option, but they could be used to encourage support of dam removal as an alternative. Finding common ground is important in divisive situations like dam repair or removal decisions. Decisionmakers, business leaders, and other concerned citizens, if asked, are all likely to agree that having a healthy river running through their community is desirable. After agreeing that a healthy river is good, commitment and consistency theories suggest that these same individuals would be more likely to later agree to consider options that could result in a healthier river, such as potentially removing the dam. Again, this exercise alone would not change social norms or ensure compliance with later requests, but it could play an important role in a successful effort to influence change toward support for sustainable ecosystem practices.

How new ideas are spread: Diffusion of innovations. In fields that have a long history of scientific outreach, such as agriculture, researchers and practitioners have spent much more time than in other fields learning how to influence the potential for adoption, and rate of adoption, of effective resource management practices, and there is a much greater acknowledgment of the role of social systems (e.g., communities) in decisionmaking. Diffusion is the spread of ideas through social systems; diffusion of innovations is the diffusion of ideas that are new, or new to the target audience (Rogers 1995). Diffusion theory grew out of rural sociology in the 1940s and has since been at the core of many university extension efforts to effect social change in agriculture, as well as behavioral change in human health and safety.

One of the most valuable contributions of diffusion research has been the identification of the different roles of individuals in encouraging adoption of new ideas. The first adopters of new ideas, so-called innovators, are typically too far ahead of the rest of the community to be helpful in encouraging change among many others. Rogers's "early adopters," how-

ever, are an integral part of the social community and are the people many others look to before using or accepting a new idea (for example, elected officials, leaders in the business community, or community elders). Early adopters help set social norms for the community. Therefore, targeting information and change efforts first to community members who have been identified (through formal or informal surveys, perhaps) as early adopters, rather than to the entire community, is an efficient method for facilitating the diffusion of information that is supportive of healthy rivers and ecosystems, because targeting saves money and speeds the process.

Social change: Techniques for changing human behaviors. Following are social science principles and concepts that hold significant potential to help change behaviors at the individual and community levels to support alternatives that could improve ecosystem health.

Social marketing. Social marketing applies marketing principles and practices to address social problems through behavioral change. It involves the marketing of a product, service, or idea where the benefit accrues not to the "seller" but to the targeted individual or society (Andreasen 1995). In social marketing, the basic means of achieving improved social welfare is to effect a change in behavior. It is much more outcome-based than many current public information and education efforts, where goals are simply to increase awareness and understanding.

Social marketing can be extremely efficient at influencing change. The individuals whose behavior one desires to change—those who have control over the outcome—are the ones who will play the primary role in the social marketing process. Unlike many other change efforts, all actions in social marketing are based on a thorough understanding of the needs, wants, and perceptions of that target audience. Social marketing begins by understanding real and perceived barriers to the desired change and then strategically delivers to key target audiences programs designed to address these concerns and to influence change (Andreasen 1995).

Social marketing practices have typically focused on individuals as the ultimate target for behavior changes such as stopping smoking, adopting certain health practices, recycling, or conserving energy, thus effectively bringing about social change, one person at a time. Social marketing practices could also hold great promise for influencing change in decisionmakers and early adopters, the opinion leaders in local communities, toward support of sustainable ecosystems.

Community-based social marketing. Community-based social marketing (CBSM) is an emerging field that is also based on psychological principles. An important distinction, however, is that CBSM makes the larger community the focus of attention, rather than individuals. It is therefore an even more efficient method for effecting social change in some cases (McKenzie-Mohr 2000).

At the heart of community-based social marketing is the identification of barriers to adopting the desired behaviors. CBSM encourages practitioners to ask three basic questions:

Box 1. Questions for researchers:**What are the potential economic impacts associated with aging dams?**

Understanding the potential economic impacts associated with keeping or removing dams is an increasingly important issue. Private and public dam owners, mayors, governors, legislators, and elected officials at all levels of government wrestle with decisions about how local communities and this country can and should deal with costs associated with aging dams. Scientific research could inform individual decisions and larger policy decisions by addressing the following questions:

- What will happen to my property value if the dam is removed? A common assumption is that property values surrounding an impoundment will decline following dam removal. Preliminary Trout Unlimited (2001) studies at one site, however, show that predicted decreases in property value had not occurred 10 years following dam removal and river restoration. Independent, scientific, peer-reviewed studies are needed to understand what short- and long-term economic impacts are associated with small dam removal, for private and public property, including businesses.
- What is the potential economic liability associated with aging dams throughout the country (or in one watershed or county)? What portion of this cost would likely be borne by taxpayers and what part by private dam owners?
- What are the costs of mitigating water quality, fisheries, and other environmental impacts associated with dams? To whom do these costs accrue, e.g., are they public or private? To whom do the benefits accrue?
- What is the cost of eliminating public safety hazards posed by dams? Who currently pays those costs, and who is likely to in the future? Considering that some states do not have dam safety programs, how accurate are assessments of dam safety costs?

What is the potential impact of the behavior? What barriers exist, real and perceived, to engaging in the desired activities? And do the resources exist to overcome identified barriers (McKenzie-Mohr 2000)?

McKenzie-Mohr's questions are used below as a framework for suggesting topics of scientific research that would be useful for informing decisionmakers (and those who influence decisionmakers) and influencing social change toward more ecologically sustainable practices.

What is the potential impact of the behavior (i.e., removing the dam in my community)? Community questions about what will happen after dam removal are wide ranging; they are societal, economic, environmental, technical, and legal in nature and would benefit from research by both natural and social scientists. The following questions, which are representative of those frequently asked by members of a community facing dam repair and removal decisions, indicate a lack of understanding of both dams and natural river systems. Will the stream dry up if the dam is removed? Will we be stuck forever with stinking mudflats? Will flooding increase without the dam (even when the dam is not a flood control structure)? Who will own the new land? Doesn't this dam have historical value? Won't the best fishing spots be lost? Won't dam removal introduce exotic or diseased species? And perhaps the most common and complex question of all is, How will keeping the dam affect my pocketbook? Our community's pocketbook? (RAW/TU 2000).

Although some of these questions may appear simple, there are no simple answers to them. The need for interdisciplinary and multidisciplinary collaboration to provide answers, or at least reasonable expectations, is evident by looking more closely at just the last question, about the economic impacts of keeping a dam or removing a dam (see box 1).

While natural science research is needed to answer many questions about potential outcomes, there is a clear need for social science research to better understand the human dimensions of dam removal.

What barriers exist, real and perceived, to engaging in the desired behavior? Identifying potential barriers calls for social science research. Experience with multiple small dam repair or removal situations suggests that barriers include lack of understanding of the values of a healthy river and how the dam may be harming the river and river life, concern about property values adjacent to the impoundment, misinformation about costs of repair and of removal, and aesthetic concerns about what the former impoundment will look like after drawdown and dam removal. These concerns represent potential barriers that could be surmounted through dissemination of scientific research findings that address the topics identified in McKenzie-Mohr's first question.

Experience repeatedly shows that other, less obvious barriers to accepting dam removal as an option also often exist; such barriers include a profound sense of loss and a sense of fear or helplessness, especially if the impetus for removal is coming from outside the community. McKenzie-Mohr (2000) and Hudson (2001) caution, however, that program planners working to effect change often mistakenly think they know what these barriers are and act accordingly, even though the perceived barriers may not reflect the actual barriers.

Social science researchers can conduct empirical research in this area and provide valuable insight into the real and perceived barriers that dissuade local community members from embracing dam removal as an alternative.

Do the resources exist to overcome identified barriers? Scientific information is needed about the resources and about the people involved, or potentially involved, in dam re-

removal decisions before the barriers to accepting dam removal as a viable alternative can be fully understood. Only when those barriers are identified can it be ascertained whether the human and financial resources exist to overcome them.

Role of scientists in decisions regarding dams and rivers

Typically, ecologists and other scientists conducting research on sustainable ecosystems focus on developing scientific principles of natural systems and practical management methods, understanding cause-and-effect relationships, and assessing environmental outcomes. Unfortunately, much of the resulting scientific information is seldom or never seen or used beyond the academic community (Doppelt 1993, Aumen and Havens 1997, Firth 1998, Lubchenco 1998).

In the case of dam removal, the problem thus far has been not so much a lack of dissemination or interpretation of scientific findings as a lack of usable scientific research findings. Although the scientific community has recognized the effects of dams on rivers for decades (Baxter 1977, Graf 1980, Petts 1980, Williams and Wolman 1984), only a handful of peer-reviewed research papers have been published on the effects of dam removals (Shuman 1995, Kanehl et al. 1997, Born et al. 1998, Bednarek 2001, Stanley et al. 2002). Only Stanley and colleagues (2002) and Kanehl and colleagues (1997) analyzed actual ecological data, and only Born and colleagues (1998) analyzed sociological data following dam removals.

As aging dams and their impacts on rivers continue to be pushed higher on the public agenda, scientists have the potential to influence whether society responds to this emerging issue as a problem or as an opportunity. Researchers today are playing a crucial role by increasingly providing data and analysis on what happens after a dam is removed. But if the goal is to have dam removal and river restoration (or other practices that lead to sustainable ecosystem health) accepted as a viable option, simply providing information may not be enough. Many citizens presume that publicly funded research will be used to benefit society, for example, to develop technology for public use and benefit. Increasingly, they also expect publicly funded research to inform public management and policy decisions—not to determine the outcome but to help understand the consequences of potential outcomes—in a manner that will benefit society (Aumen and Havens 1997, Lubchenco 1998, Norton 1998, Bjorkland and Pringle 2001, Hudson 2001). In Wisconsin, for example, there is a long history of interaction between university researchers and decisionmakers for the purpose of informing public policy. The “Wisconsin Idea” is well known in that state, where, historically, such interactions are strongly encouraged and facilitated (University of Wisconsin—Extension 1981).

Human behavioral change is central to the notion of ecological sustainability, and high levels of public support for such sustainability—which are not currently present—will be needed (Orr 1992, McKenzie-Mohr 2000, Bjorkland and Pringle 2001). Ecologists and other scientists who want their

research to result in healthier ecosystems hope for change. Such change will require elected officials and other decisionmakers, including resource agency personnel, industry representatives, and private citizens, to step outside their comfort zone to accept new ideas. Scientists who want to help facilitate social change to benefit ecosystem health may need to stretch the boundaries of their own comfort zones as well.

We are not suggesting by any means that all scientists forgo conducting “pure” research. Objective information is critical to the credibility of public education efforts regarding natural resource issues, especially controversial ones (Johnson and Jacobs 1994). We are suggesting, however, that some scientists consider using their knowledge to address practical management challenges and to inform public policy on dams and rivers by ensuring that their scientific findings are interpreted and disseminated beyond the academic community (see figure 1). We hope that yet others will offer their technical expertise directly to resource agencies and change agents, such as university extension and resource agency personnel, as well as to conservation organizations and dam owners themselves. It is possible to inform without advocating (Blockstein 2002).

Figure 1. Potential roles of scientists in influencing social change around dams and rivers.

Conduct pure research

Conduct pure research on the environmental, economic, and societal impacts of dams and dam removal, and publish findings.



Conduct applied research

Allow practitioner needs to shape research agendas. Address research topics that could directly assist dam repair or removal decisions and dam removal management.



Disseminate findings

Actively facilitate the interpretation and dissemination of scientific information to resource agencies, communities, decisionmakers, conservation organizations, and dam owners, through presentations and publications outside the academic community.



Provide technical expertise

Provide professional technical expertise on potential outcomes of various alternatives through public comment processes or as a technical advisor to a resource agency, conservation organization, or professional association.



Actively promote findings

Write letters to the editor, elected officials, agency heads, and other decisionmakers and influential leaders.

Scientists who believe that science should inform public management and policy decisions, and who agree that societal change toward more sustainable ecosystem practices is necessary, are the best candidates for successfully moving from conducting pure research to actively promoting their findings.

Conclusion

Significant changes in human behavior are required to achieve sustainable ecosystem practices. A key component in achieving desired ecological outcomes will be to improve the process of decisionmaking so that alternatives that could improve ecosystem health, such as selective dam removal, are considered and accepted or rejected on their own merits. Scientists, especially social scientists, can play important roles in enabling such social change by conducting applied research and working to interpret and disseminate findings to decisionmakers.

Acknowledgments

The authors would like to thank David Hart of the Academy of Natural Sciences for recognizing the value of human dimensions in dam removal issues and the value of practitioner expertise, and for his encouragement and comments on earlier versions of this paper. Thanks also to two anonymous reviewers for their suggestions. Finally, we extend our thanks to The Pew Charitable Trusts for support of this research.

References cited

- [ACA] American Canoeing Association. 2000. Utilizing the economics of paddling. (27 June 2002; www.acanet.org/acanet.htm)
- [AR] American Rivers. 2001. Dams removed in 1999, 2000, 2001. (27 June 2002; www.amrivers.org/damremoval/damremovals2001.htm)
- [AR/FE/TU] American Rivers, Friends of the Earth, Trout Unlimited. 1999. Dam Removal Success Stories: Restoring Rivers through Selective Removal of Dams That Don't Make Sense. Washington (DC): AR/FE/TU.
- [AR/TU] American Rivers, Trout Unlimited. 2002. Exploring Dam Removal: A Decision-Making Guide.
- [ASCE] American Society of Civil Engineers. 2001. 2001 report card for America's infrastructure. (27 June 2002; www.asce.org/reportcard)
- Andreasen AR. 1995. Marketing Social Change. San Francisco: Jossey-Bass.
- Andrews ED. 1986. Downstream effects of Flaming Gorge Reservoir on the Green River, Colorado and Utah. *Geological Society of America Bulletin* 97: 1012–1023.
- [ASDSO] Association of State Dam Safety Officials. 2001. Dam safety: A national concern. (27 June 2002; www.damsafety.org/infocentral.cfm?content=damsafety101)
- Aumen N, Havens K. 1997. Needed: A cadre of applied scientists skilled in basic science, communication, and aquatic resource management. *Journal of the North American Benthological Society* 16: 710–716.
- Baxter RM. 1977. Environmental effects of dams and impoundments. *Annual Review of Ecology and Systematics* 8: 255–283.
- Bednarek AT. 2001. Undamming rivers: A review of the ecological impacts of dam removal. *Environmental Management* 27: 803–814.
- Bjorkland R, Pringle CM. 2001. Educating our communities and ourselves about conservation of aquatic resources through environmental outreach. *BioScience* 51: 279–282.
- Blockstein DE. 2002. How to lose your political virginity while keeping your scientific credibility. *BioScience* 52: 91–96.
- Born S, Genskow K, Filbert T, Hernandez-Mora N, Keefer M, White K. 1998. Socioeconomic and institutional dimensions of dam removals: The Wisconsin experience. *Environmental Management* 22: 359–370.
- Brandt SA. 2000. Classification of geomorphological effects downstream of dams. *Catena* 40: 375–401.
- Chien N. 1985. Changes in river regime after the construction of upstream reservoirs. *Earth Surface Processes and Landforms* 10: 143–159.
- Cialdini R. 2001. *Influence: Science and Practice*. Needham Heights (MA): Allyn and Bacon.
- Doppelt R. 1993. The vital role of the scientific community in new river conservation strategies. *Journal of Personality and Social Psychology* 12: 189–193.
- Fedler AJ. 2000. *Participating in Boating and Fishing: A Literature Review*. Alexandria (VA): Recreational Boating and Fishing Foundation.
- [FEMA] Federal Emergency Management Agency. 1999. *Availability of Dam Insurance: A Report to Congress*. Washington (DC): National Dam Safety Program, FEMA Mitigation Directorate.
- Firth P. 1998. Freshwater: Perspectives on the integration of research, education, and decision making. *Ecological Applications* 8: 601–609.
- Freedman J, Fraser S. 1966. Compliance without pressure: The foot-in-the-door technique. *Journal of Personality and Social Psychology* 4: 195–203.
- Graf WL. 1980. The effect of dam closure on downstream rapids. *Water Resources Research* 16: 129–136.
- . 2001. Damage control: Restoring the physical integrity of America's rivers. *Annals of the Association of American Geographers* 91: 1–27.
- Hadley RF, Emmett WW. 1998. Channel changes downstream from a dam. *Journal of the American Water Resources Association* 34: 629–637.
- Hudson SJ. 2001. Challenges for environmental education: Issues and ideas for the 21st century. *Bioscience* 51: 283–288.
- Johnson SE, Jacobs HM. 1994. Public education for growth management: Lessons from Wisconsin's farmland preservation program. *Journal of Soil and Water Conservation* 49: 333–338.
- Johnston LR, Associates. 1992. *A Status Report on the Nation's Floodplain Management Activity: An Interim Report*. Washington (DC): Federal Interagency Floodplain Management Task Force. Contract no. TV-72105A.
- Kanehl PD, Lyons J, Nelson JE. 1997. Changes in the habitat and fish community of the Milwaukee river, Wisconsin, following removal of the Woolen Mills Dam. *North American Journal of Fisheries Management* 17: 387–398.
- Katzev R, Wang T. 1994. Can commitment change behavior? A case study of environmental actions. *Journal of Social Behavior and Personality* 9: 13–26.
- Ligon FK, Dietrich WE, Trush WJ. 1995. Downstream ecological effects of dams. *BioScience* 45: 183–192.
- Lubchenco J. 1998. Entering the century of the environment: A new social contract for science. *Science* 279: 491–497.
- Magilligan FJ, Nislow KH. 2001. Long-term changes in regional hydrologic regime following impoundment in a humid-climate watershed. *Journal of the American Water Resources Association* 37: 1557–1570.
- McKenzie-Mohr D. 2000. Promoting sustainable behavior: An introduction to community-based social marketing. *Journal of Social Issues* 56: 543–554.
- . 2002. *Fostering sustainable behavior*. (28 June 2002; www.cbsn.co.m)
- McKenzie-Mohr D, Smith W. 1999. *Fostering Sustainable Behavior: Community-Based Social Marketing*. Gabriola Island (Canada): New Society Publishers.
- [NID] National Inventory of Dams. 2001. (28 June 2002; <http://crunch.tec.army.mil/nid/webpages/nid.cfm>)
- Norton B. 1998. Improving ecological communication: The role of ecologists in environmental policy formation. *Ecological Applications* 8: 350–364.
- Orr D. 1992. *Ecological Literacy*. Albany (NY): SUNY Press.
- Petts GE. 1980. Long-term consequences of upstream impoundment. *Environmental Conservation* 7: 325–332.
- Petts GE, Pratts JD. 1983. Channel changes following reservoir construction on a lowland river. *Catena* 10: 77–85.

- Power ME, Dietrich WE, Finlay JC. 1996. Dams and downstream aquatic biodiversity: Potential food web consequences of hydrologic and geomorphic change. *Environmental Management* 20: 887–895.
- Rogers EM. 1995. *Diffusion of Innovations*. New York: Free Press.
- [RAW/TU] River Alliance of Wisconsin, Trout Unlimited. 2000. *Dam Removal: A Citizen's Guide to Restoring Rivers*. Madison (WI): RAW/TU.
- Shuman JR. 1995. Environmental considerations for assessing dam removal alternatives for river restoration. *Regulated Rivers: Research and Management* 11: 249–261.
- Stanley EH, Luebke MA, Doyle MW, Marshall DW. 2002. Short-term changes in channel form and macroinvertebrate communities following low-head dam removal. *Journal of the North American Benthological Society* 21: 172–187.
- [TU] Trout Unlimited. 2001. *Small Dam Removal: A Review of Potential Economic Benefits*. Arlington (VA): Trout Unlimited.
- University of Wisconsin–Extension. 1981. *The Wisconsin Idea: A Tribute to Carlisle P. Runge*, Proceedings of the Colloquium, March 28, 1981. Madison: University of Wisconsin–Extension.
- Williams GP, Wolman MG. 1984. *Downstream Effects of Dams on Alluvial Rivers*. Washington (DC): US Geological Survey. Geological Survey Professional Paper 1286.

Centre d'expertise
hydrique

Québec



Ministère du Développement durable, de l'Environnement et des Parcs

Côte-Nord (09) - Liste des barrages

No. du barrage ▲	Nom du barrage	Municipalité	Provc	Lac et cours d'eau
—	—	Tadoussac	La Haute-Côte-Nord	Moulin à Baude, Rivière du
—	—	Tadoussac	La Haute-Côte-Nord	AQUEDUC (DE L')
—	Anse-à-l'Eau-3, Barrage de l'	Tadoussac	La Haute-Côte-Nord	ANSE A L'EAU (DE L')
—	Anse-à-l'Eau-1, Barrage de l'	Tadoussac	La Haute-Côte-Nord	Trib.48D.08M.27S.-69D.43M.31S.
—	—	Sacré-Coeur	La Haute-Côte-Nord	Chute à Brûlé, Ruisseau de la
—	—	Sacré-Coeur	La Haute-Côte-Nord	F2309
—	—	Sacré-Coeur	La Haute-Côte-Nord	Chute à Brûlé, Ruisseau de la
—	—	Les Bergeronnes	La Haute-Côte-Nord	F1627
—	—	Les Bergeronnes	La Haute-Côte-Nord	—
—	Sables-inférieurs, Barrage des	Les Bergeronnes	La Haute-Côte-Nord	Petites Bergeronnes, Rivière des
—	—	Les Escoumins	La Haute-Côte-Nord	PARENT
—	—	Les Escoumins	La Haute-Côte-Nord	Escoumins, Rivière des
—	—	Longue-Rive	La Haute-Côte-Nord	Portneuf, Rivière
—	—	Portneuf-sur-Mer	La Haute-Côte-Nord	Portneuf, Rivière
—	—	Portneuf-sur-Mer	La Haute-Côte-Nord	Portneuf, Rivière
—	—	Forestville	La Haute-Côte-Nord	Sault aux Cochons, Rivière du
—	Sault-aux-Cochons, Barrage du	Forestville	La Haute-Côte-Nord	Sault aux Cochons, Rivière du
—	Bersimis-2, Digue 2	Lac-au-Brochet	La Haute-Côte-Nord	—
—	Bersimis-2, Barrage	Lac-au-Brochet	La Haute-Côte-Nord	—
—	Bersimis-2, Remblai rive gauche	Lac-au-Brochet	La Haute-Côte-Nord	—
—	Desroches, Barrage	Lac-au-Brochet	La Haute-Côte-Nord	Pipmuacan, Réervoir
—	Betsiamites, Barrage	Lac-au-Brochet	La Haute-Côte-Nord	Pipmuacan, Réervoir

	Bersimis-1, Barrage	Lac-au-Brochet	La Haute-Côte-Nord	Pipmuacan, Réervoir
	—	Lac-au-Brochet	La Haute-Côte-Nord	Petits Escoumins, Lac des
	—	Lac-au-Brochet	La Haute-Côte-Nord	Petits Escoumins, Lac des
	Maclure, Barrage	Lac-au-Brochet	La Haute-Côte-Nord	Maclure, Lac
	Chatignies, Barrage	Lac-au-Brochet	La Haute-Côte-Nord	CHATIGNIES
	—	Lac-au-Brochet	La Haute-Côte-Nord	MONGRAIN
	—	Lac-au-Brochet	La Haute-Côte-Nord	PETITE MONTAGNE (DE LA)
	Boucher, Barrage	Lac-au-Brochet	La Haute-Côte-Nord	Boucher, Lac
	Pipe, Barrage à la	Lac-au-Brochet	La Haute-Côte-Nord	PIPE (À LA)
	Gorgotton, Barrage	Lac-au-Brochet	La Haute-Côte-Nord	Gorgotton, Lac
	—	Lac-au-Brochet	La Haute-Côte-Nord	Coeurs, Lac des
	Brûlé, Barrage	Lac-au-Brochet	La Haute-Côte-Nord	Brûlé, Lac
	Piliers, Barrage des	Lac-au-Brochet	La Haute-Côte-Nord	PILIER (DES)
	Régis, Barrage	Lac-au-Brochet	La Haute-Côte-Nord	REGIS
	Lapointe, Barrage	Lac-au-Brochet	La Haute-Côte-Nord	LAPOINTE
	—	Lac-au-Brochet	La Haute-Côte-Nord	DOW
	—	Lac-au-Brochet	La Haute-Côte-Nord	SUZANNE
	Marot, Barrage	Lac-au-Brochet	La Haute-Côte-Nord	NORD (GRAND, DU)
	Griffiths, Barrage	Lac-au-Brochet	La Haute-Côte-Nord	GRIFFITHS
	—	Lac-au-Brochet	La Haute-Côte-Nord	PORTAGE (GRAND, DU)
	—	Lac-au-Brochet	La Haute-Côte-Nord	Sault aux Cochons, Rivière du
	—	Lac-au-Brochet	La Haute-Côte-Nord	Kakuskanus, Lac
	Éphrem, Barrage	Lac-au-Brochet	La Haute-Côte-Nord	ÉPHREM
	Grand-Canyon, Barrage du	Lac-au-Brochet	La Haute-Côte-Nord	CASSANDRE
	Croche, Barrage	Lac-au-Brochet	La Haute-Côte-Nord	CROCHE
	—	Lac-au-Brochet	La Haute-Côte-Nord	ISIDORE
	Polette, Barrage	Lac-au-Brochet	La Haute-Côte-Nord	Polette, Lac
	Grand-Lac, Barrage du	Lac-au-Brochet	La Haute-Côte-Nord	Grand Lac, Le
	Nipi, Barrage	Lac-au-Brochet	La Haute-Côte-Nord	Nipi, Rivière
	Fries, Barrage	Baie-Trinité	Manicouagan	FRIES
	Jacques, Barrage	Baie-Trinité	Manicouagan	JACQUES
	Mitchell, Barrage	Baie-Trinité	Manicouagan	MITCHELL

	Ladoga, Barrage	Baie-Trinité	Manicouagan	LADOGA
	Godfrey, Barrage	Baie-Trinité	Manicouagan	GODFREY
	Baie-Trinité, Barrage de	Baie-Trinité	Manicouagan	—
	Raquette, Barrage à la	Godbout	Manicouagan	RAQUETTE (A LA)
	Île, Barrage à l'	Godbout	Manicouagan	Île, Lac à l'
	—	Franquelin	Manicouagan	MORRIS
	Oignon, Barrage à l'	Franquelin	Manicouagan	OIGNON (A L')
	Longchamp, Barrage	Franquelin	Manicouagan	Longchamp, Lac
	Amédée, Barrage	Baie-Comeau	Manicouagan	Amédée, Rivière
	—	Baie-Comeau	Manicouagan	CHASSE (A LA)
	McCormick, Barrage	Baie-Comeau	Manicouagan	Manicouagan, Rivière
	McCormick, Centrale	Baie-Comeau	Manicouagan	Manicouagan, Rivière
	Couillard, Barrage	Baie-Comeau	Manicouagan	COUILLARD
	—	Baie-Comeau	Manicouagan	La Chesnaye, Lac
	Rivière-aux- Anglais, Barrage de la	Baie-Comeau	Manicouagan	RIVIERE AUX ANGLAIS (DE LA)
	—	Baie-Comeau	Manicouagan	CASTELNAU
	—	Baie-Comeau	Manicouagan	COMEAU
	Manic-1, Barrage	Baie-Comeau	Manicouagan	Manicouagan, Rivière
	Manic-2, Barrage	Baie-Comeau	Manicouagan	Manicouagan, Rivière
	Manic-3, Barrage	Rivière-aux- Outardes	Manicouagan	Manicouagan, Rivière
	Manic-3, Barrage secondaire	Rivière-aux- Outardes	Manicouagan	Manicouagan, Rivière
	Outardes-2, Auxiliaire no. 1 aux	Chute-aux- Outardes	Manicouagan	—
	Outardes-2, Auxiliaire no.2 aux	Pointe-aux- Outardes	Manicouagan	—
	Outardes-2, Auxiliaire no. 3 aux	Chute-aux- Outardes	Manicouagan	—
	Outardes-2, Auxiliaire no. 4 aux	Chute-aux- Outardes	Manicouagan	—
	Outardes-2, Auxiliaire no. 5 aux	Chute-aux- Outardes	Manicouagan	—
	Outardes-2, Barrage aux	Ragueneau	Manicouagan	Outardes, Rivière aux
	Outardes-2, Digue Est aux	Pointe-aux- Outardes	Manicouagan	Outardes, Rivière aux
	Outardes-2, Digue du Ruisseau Est aux	Chute-aux- Outardes	Manicouagan	—
	—	Rivière-aux- Outardes	Manicouagan	Loutre, Lac à la
	Brisson, Barrage	Rivière-aux- Outardes	Manicouagan	Brisson, Rivière
	Thérèse, Barrage	Rivière-aux- Outardes	Manicouagan	THERESE

	Limite-3, Barrage de la	Rivière-aux-Outardes	Manicouagan	LIMITE (DE LA)
	Limite-2, Barrage de la	Rivière-aux-Outardes	Manicouagan	—
	Limite-1, Barrage de la	Rivière-aux-Outardes	Manicouagan	Trib.49D.19M.00S.-68D.27M.00S.
	Rambois, Barrage	Rivière-aux-Outardes	Manicouagan	RAMBOIS
	Arthur, Barrage	Rivière-aux-Outardes	Manicouagan	ARTHUR
	Robot, Barrage du	Rivière-aux-Outardes	Manicouagan	Robot, Lac du
	Varin, Barrage	Rivière-aux-Outardes	Manicouagan	VARIN
	Franquelin, Barrage	Rivière-aux-Outardes	Manicouagan	—
	—	Rivière-aux-Outardes	Manicouagan	Franquelin, Lac
	—	Rivière-aux-Outardes	Manicouagan	MC CORMICK
	Pistucanis, Barrage	Rivière-aux-Outardes	Manicouagan	PISTUCANIS
	—	Rivière-aux-Outardes	Manicouagan	F1896
	—	Rivière-aux-Outardes	Manicouagan	F1897
	—	Rivière-aux-Outardes	Manicouagan	F1898
	Brûlé, Barrage	Rivière-aux-Outardes	Manicouagan	BRULE
	Perreault, Barrage	Rivière-aux-Outardes	Manicouagan	—
	Outardes-4, Évacuateur aux	Rivière-aux-Outardes	Manicouagan	Outardes, Rivière aux
	Outardes-4, Barrage aux	Rivière-aux-Outardes	Manicouagan	Outardes, Rivière aux
	Outardes-4, Remblai no. 2 aux	Rivière-aux-Outardes	Manicouagan	Outardes, Rivière aux
	Outardes-4, Remblai no.3 aux	Rivière-aux-Outardes	Manicouagan	Outardes, Rivière aux
	Outardes-4, Remblai no.4 aux	Rivière-aux-Outardes	Manicouagan	Outardes, Rivière aux
	Outardes-4, Remblai no. 6 aux	Rivière-aux-Outardes	Manicouagan	Outardes, Rivière aux
	Outardes-4, Remblai no. 7A aux	Rivière-aux-Outardes	Manicouagan	Outardes, Rivière aux
	Outardes-4, Remblai no. 7B aux	Rivière-aux-Outardes	Manicouagan	Outardes, Rivière aux
	Outardes-4, Remblai no. 8A aux	Rivière-aux-Outardes	Manicouagan	Outardes, Rivière aux
	Outardes-4, Remblai no. 8B aux	Rivière-aux-Outardes	Manicouagan	Outardes, Rivière aux
	Outardes-3, Évacuateur de	Rivière-aux-Outardes	Manicouagan	Outardes, Rivière aux

	crues aux			
	Outardes-3, Barrage aux	Rivière-aux-Outardes	Manicouagan	Outardes, Rivière aux
	Outardes-3, Digue latérale aux	Rivière-aux-Outardes	Manicouagan	Outardes, Rivière aux
	Lac-Sainte-Anne, Remblai Sud-Est du	Rivière-aux-Outardes	Manicouagan	SAINTE-ANNE
	Manic-5-PA, Barrage	Rivière-aux-Outardes	Manicouagan	Manicouagan, Réservoir
	Daniel-Johnson, Barrage	Rivière-aux-Outardes	Manicouagan	Manicouagan, Réservoir
	Manic-5, Prise d'eau	Rivière-aux-Outardes	Manicouagan	Manicouagan, Réservoir
	Manic-5, Évacuateur principal	Rivière-aux-Outardes	Manicouagan	Manicouagan, Réservoir
	Manic-5, Remblai rive droite	Rivière-aux-Outardes	Manicouagan	Manicouagan, Réservoir
	Manic-5, Remblai rive gauche	Rivière-aux-Outardes	Manicouagan	Manicouagan, Réservoir
	—	Sept-Îles	Sept-Rivières	Wabush, Réservoir
	Sainte-Marguerite-2, Barrage	Lac-Walker	Sept-Rivières	Sainte-Marguerite, Rivière
	—	Sept-Îles	Sept-Rivières	Sainte-Marguerite, Rivière
	Rivière-Riverin, Barrage de la	Port-Cartier	Sept-Rivières	Riverin, Rivière
	Lapointe, Barrage	Rivière-Mouchalagane	Caniapiscau	Hart Jaune, Rivière
	Hart-Jaune, Barrage secondaire	Rivière-Mouchalagane	Caniapiscau	Hart Jaune, Rivière
	Hart-Jaune, Digue HJ-3	Rivière-Mouchalagane	Caniapiscau	Hart Jaune, Rivière
	Hart-Jaune, Barrage principal	Rivière-Mouchalagane	Caniapiscau	Hart Jaune, Rivière
	Hart-Jaune, Digue HJ-2	Rivière-Mouchalagane	Caniapiscau	Hart Jaune, Rivière
	Hart-Jaune, Digue HJ-1	Rivière-Mouchalagane	Caniapiscau	Hart Jaune, Rivière
	Hart-Jaune, Barrage prise d'eau de la	Rivière-Mouchalagane	Caniapiscau	Hart Jaune, Rivière
	Hart-Jaune, Digue HJ-8	Rivière-Mouchalagane	Caniapiscau	Hart Jaune, Rivière
	Hart-Jaune, Digue HJ-7	Rivière-Mouchalagane	Caniapiscau	Hart Jaune, Rivière
	Hart-Jaune, Barrage intermédiaire de la	Rivière-Mouchalagane	Caniapiscau	Hart Jaune, Rivière
	Hart-Jaune, Digue HJ-9	Rivière-Mouchalagane	Caniapiscau	Manicouagan, Petit lac
	Hart-Jaune, Barrage de la	Rivière-Mouchalagane	Caniapiscau	Manicouagan, Petit lac
	Caniapiscau, Digue KC-23A	Rivière-Mouchalagane	Caniapiscau	Caniapiscau, Lac
	Caniapiscau, Digue KC-25A	Caniapiscau	Caniapiscau	Caniapiscau, Lac

	Caniapiscau, Digue KA-02	Caniapiscau	Caniapiscau	Caniapiscau, Lac
	Caniapiscau, Digue KA-03	Caniapiscau	Caniapiscau	Caniapiscau, Lac
	Caniapiscau, Digue KA-04	Caniapiscau	Caniapiscau	Caniapiscau, Lac
	Caniapiscau, Digue KA-05	Caniapiscau	Caniapiscau	Caniapiscau, Lac
	Caniapiscau, Digue KA-06	Caniapiscau	Caniapiscau	Caniapiscau, Lac
	Caniapiscau, Digue KA-06A	Caniapiscau	Caniapiscau	Caniapiscau, Lac
	Caniapiscau, Digue KA-01	Caniapiscau	Caniapiscau	Caniapiscau, Lac
	Magpie, Barrage de	Rivière-Saint- Jean	Minganie	Magpie, Rivière
	Lac-Robertson, Digue de protection latérale du	Gros-Mécatina	Côte-Nord-du- Golfe-Saint- Laurent	—
	Lac-Robertson, Évacuateur du	Gros-Mécatina	Côte-Nord-du- Golfe-Saint- Laurent	—
	Lac-Robertson, Barrage voute du	Gros-Mécatina	Côte-Nord-du- Golfe-Saint- Laurent	—
	Lac-Robertson, Barrage du	Petit-Mécatina	Minganie	—
	Caniapiscau, Digue KC-26	Caniapiscau	Caniapiscau	—
	Caniapiscau, Digue KC-28A	Caniapiscau	Caniapiscau	—
	Caniapiscau, Digue KC-28-B	Caniapiscau	Caniapiscau	—
	Caniapiscau, Digue KC-29	Caniapiscau	Caniapiscau	—
	Caniapiscau, Digue KC-30	Caniapiscau	Caniapiscau	—
	Anse-à-l'Eau-2, Barrage de l'	Tadoussac	La Haute-Côte- Nord	Trib.48D.08M.27S.- 69D.43M.31S.
	Pionniers, Barrage des	Port-Cartier	Sept-Rivières	Rochers, Rivière aux
	—	L'Île-d'Anticosti	Minganie	SAINT-GEORGES
	Lac-Faure, Barrage du	L'Île-d'Anticosti	Minganie	Faure, Lac
	Nicette, Barrage	Lac-au-Brochet	La Haute-Côte- Nord	—
	—	Port-Cartier	Sept-Rivières	Riverin, Rivière
	—	Baie-Comeau	Manicouagan	—
	Tremblay, Barrage	Betsiamites	Manicouagan	TREMBLAY
	Fèves, Barrage des	Lac-au-Brochet	La Haute-Côte- Nord	FEVES (DES)
	Premier-Lac-de- la-Rivière- Rocheuse, Barrage du	Lac-au-Brochet	La Haute-Côte- Nord	Rocheuse, Rivière
	Bacagnole-1, Barrage du	Lac-au-Brochet	La Haute-Côte- Nord	BACAGNOL (DU)
	Sault-aux- Cochons, Digue Nord-Est du	Lac-au-Brochet	La Haute-Côte- Nord	Sault aux Cochons, Lac du

	Sault-aux-Cochons, Dérivation partielle du	Lac-au-Brochet	La Haute-Côte-Nord	Sault aux Cochons, Lac du
	Toulnustouc, Barrage de la	Rivière-aux-Outardes	Manicouagan	—
	Toulnustouc, Digue Sud de la	Rivière-aux-Outardes	Manicouagan	—
	Toulnustouc, Seuil aux crans serrés de la	Rivière-aux-Outardes	Manicouagan	Toulnustouc, Rivière
	—	Rivière-aux-Outardes	Manicouagan	—
	Perles, Barrage aux	Lac-au-Brochet	La Haute-Côte-Nord	Canards, Rivière aux
	—	Lac-au-Brochet	La Haute-Côte-Nord	CORMIER
	—	Les Bergeronnes	La Haute-Côte-Nord	Sables, Lac des
	—	Les Bergeronnes	La Haute-Côte-Nord	PARADIS
	Petite-Rivière-Sainte-Marguerite, Barrage de la	Sept-Îles	Sept-Rivières	Sainte-Marguerite, Petite rivière
	Denis-Perron, Barrage	Lac-Walker	Sept-Rivières	—
	Sainte-Marguerite-3, Évacuateur de la	Lac-Walker	Sept-Rivières	—
	Sainte-Marguerite-3, Seuil écologique de la	Lac-Walker	Sept-Rivières	—
	—	Schefferville	Caniapiscau	KNOB
	—	Rivière-aux-Outardes	Manicouagan	SHACKLETON NO 3*
	—	Sacré-Coeur	La Haute-Côte-Nord	—
	—	Sept-Îles	Sept-Rivières	Sainte-Marguerite, Rivière
	—	Lac-Walker	Sept-Rivières	Cacaoui, Lac
	—	Lac-Walker	Sept-Rivières	Cacaoui, Lac
	—	Baie-Comeau	Manicouagan	CASTELNAU
	—	Baie-Comeau	Manicouagan	CASTELNAU
	—	Baie-Comeau	Manicouagan	CASTELNAU
	—	Baie-Comeau	Manicouagan	CASTELNAU
	—	Baie-Comeau	Manicouagan	CASTELNAU
	—	Baie-Comeau	Manicouagan	CASTELNAU
	—	L'Île-d'Anticosti	Minganie	SAINT-GEORGES

Aide

Nouvelle recherche

Pour nous joindre

« — » : le tiret indique que le renseignement n'était pas disponible au moment de la dernière mise à jour ou n'existe pas. C'est notamment le cas lorsque le barrage ou le cours d'eau n'a pas de désignation officielle à la

« » : possibilité de trier le résultat en appuyant sur le titre de la colonne.

Québec

09 - La Côte-Nord ainsi que ses municipalités régionales de comté (MRC) et territoire équivalent (TE)

Fiche synthèse par MRC ou par TE

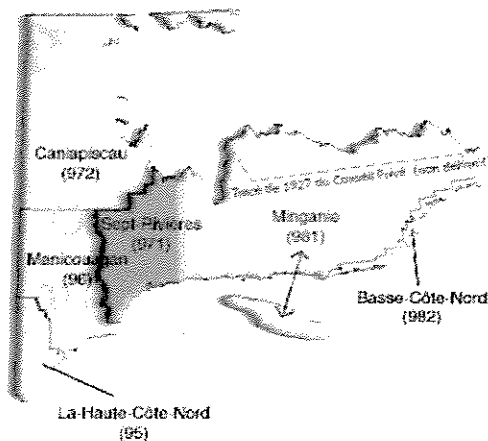
Positionnez le curseur sur une MRC ou sur un TE de la carte pour afficher sa fiche synthèse.

Profils de la région administrative

- Commerce extérieur
- Comptes économiques
- Conditions de vie et bien-être
- Construction et habitation
- Culture et communications
- Démographie
- Éducation
- Industrie bioalimentaire
- Investissements
- Santé
- Science, technologie et innovation
- Travail et rémunération

Profils comparatifs des MRC et du TE

- Comptes économiques
- Conditions de vie et bien-être
- Construction et habitation
- Démographie
- Déplacements domicile-travail
- Travail et rémunération



Sept-Rivières

Superficie en terre ferme (2007)	29 946 km2
Densité de population (2007)	1,2 hab/km2
Population totale (2007)	34 702 hab.
0-14 ans	6 136 hab.
15-24 ans	4 216 hab.
25-44 ans	9 938 hab.
45-64 ans	10 175 hab.
65 ans et plus	4 237 hab.
Solde migratoire interrégional (2006-2007)	-63 hab.
Perspectives démographiques (variation 2026/2001)	- 16,3 %
Travailleurs de 25-64 ans (2005)	14 415
Taux de travailleurs de 25-64 ans (2005)	72,3 %
Revenu d'emploi moyen des travailleurs de 25-64 ans (2005)	44 603 \$
Taux d'assistance-emploi (2006)	6,5 %
Taux de faible revenu des familles (2005)	8,4 %
Revenu personnel disponible par habitant (2007)	27 046 \$
Valeur totale des permis de bâtir (2007)	41 882 k\$
Valeur foncière moyenne des maisons unifamiliales (2008)	106 214 \$

Pour imprimer la fiche synthèse d'une MRC ou d'un TE, Choisissez une MRC ou un TE.

Pour mieux connaître le Système du code géographique du Québec, consultez la [Base de données du code géographique](#).

Haut de la page

09 - La Côte-Nord ainsi que ses municipalités régionales de comté (MRC) et territoire équivalent (TE)

Fiche synthèse par MRC ou par TE

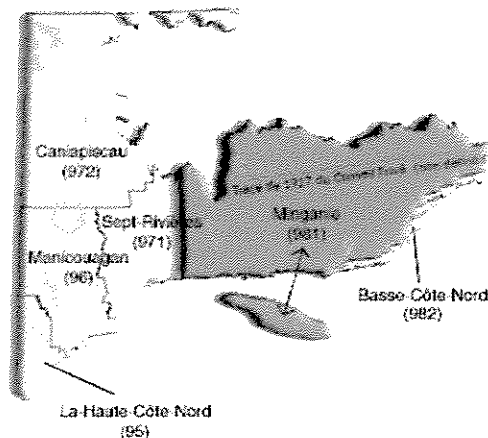
Positionnez le curseur sur une MRC ou sur un TE de la carte pour afficher sa fiche synthèse.

Profils de la région administrative

- Commerce extérieur
- Comptes économiques
- Conditions de vie et bien-être
- Construction et habitation
- Culture et communications
- Démographie
- Éducation
- Industrie bioalimentaire
- Investissements
- Santé
- Science, technologie et innovation
- Travail et rémunération

Profils comparatifs des MRC et du TE

- Comptes économiques
- Conditions de vie et bien-être
- Construction et habitation
- Démographie
- Déplacements domicile-travail
- Travail et rémunération



Minganie

Superficie en terre ferme (2007)	90 997 km ²
Densité de population (2007)	0,1 hab./km ²
Population totale (2007)	6 504 hab.
0-14 ans	1 154 hab.
15-24 ans	842 hab.
25-44 ans	1 712 hab.
45-64 ans	2 017 hab.
65 ans et plus	779 hab.
Solde migratoire interrégional (2006-2007)	-47 hab.
Perspectives démographiques (variation 2026/2001)	-16,8 %
Travailleurs de 25-64 ans (2005)	2 338
Taux de travailleurs de 25-64 ans (2005)	63,5 %
Revenu d'emploi moyen des travailleurs de 25-64 ans (2005)	36 363 \$
Taux d'assistance-emploi (2006)	4,2 %
Taux de faible revenu des familles (2005)	13,2 %
Revenu personnel disponible par habitant (2007)	23 476 \$
Valeur totale des permis de bâtir (2007)	15 188 k\$
Valeur foncière moyenne des maisons unifamiliales (2008)	71 381 \$

Pour imprimer la fiche synthèse d'une MRC ou d'un TE, Choisissez une MRC ou un TE.

Pour mieux connaître le Système du code géographique du Québec, consultez la [Base de données du code géographique](#).

Haut de la page

09 - La Côte-Nord ainsi que ses municipalités régionales de comté (MRC) et territoire équivalent (TE)

Fiche synthèse par MRC ou par TE

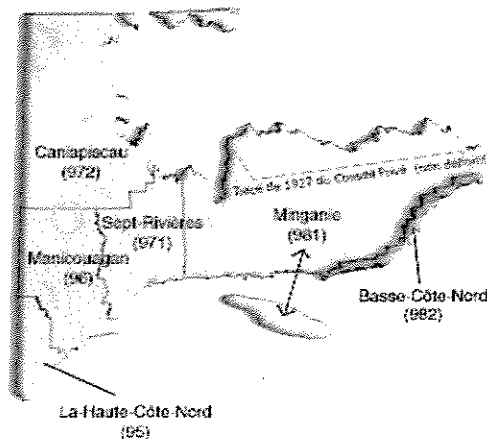
Positionnez le curseur sur une MRC ou sur un TE de la carte pour afficher sa fiche synthèse.

Profils de la région administrative

- Commerce extérieur
- Comptes économiques
- Conditions de vie et bien-être
- Construction et habitation
- Culture et communications
- Démographie
- Éducation
- Industrie bioalimentaire
- Investissements
- Santé
- Science, technologie et innovation
- Travail et rémunération

Profils comparatifs des MRC et du TE

- Comptes économiques
- Conditions de vie et bien-être
- Construction et habitation
- Démographie
- Déplacements domicile-travail
- Travail et rémunération



Basse-Côte-Nord

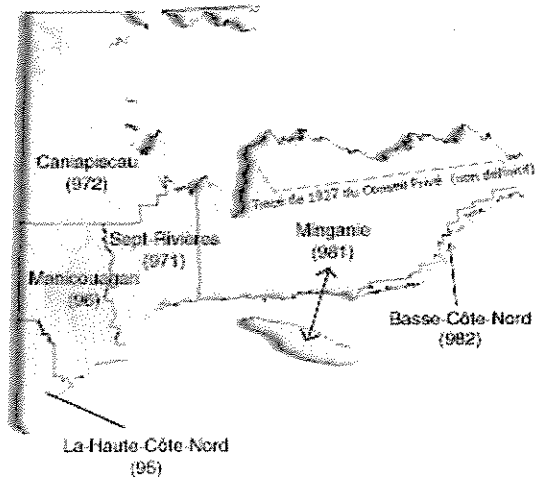
Superficie en terre ferme (2007)	5 528 km ²
Densité de population (2007)	1,0 hab/km ²
Population totale (2007)	5 613 hab.
0-14 ans	1 056 hab.
15-24 ans	793 hab.
25-44 ans	1 681 hab.
45-64 ans	1 423 hab.
65 ans et plus	661 hab.
Solde migratoire interrégional (2006-2007)	-68 hab.
Perspectives démographiques (variation 2026/2001)	- 15,0 %
Travailleurs de 25-64 ans (2005)	1 696
Taux de travailleurs de 25-64 ans (2005)	54,1 %
Revenu d'emploi moyen des travailleurs de 25-64 ans (2005)	30 996 \$
Taux d'assistance-emploi (2006)	2,9 %
Taux de faible revenu des familles (2005)	20,2 %
Revenu personnel disponible par habitant (2007)	17 446 \$
Valeur totale des permis de bâtir (2007)	-
Valeur foncière moyenne des maisons unifamiliales (2008)	48 729 \$

Pour imprimer la fiche synthèse d'une MRC ou d'un TE, Choisissez une MRC ou un TE.

Pour mieux connaître le Système du code géographique du Québec, consultez la [Base de données du code géographique](#).

Haut de la page

Institut de la statistique du Québec



09 – Côte-Nord

Superficie en terre ferme (2007)	236 699,6 km ²
Densité de population (2007)	0,4 hab./km ²
Population totale (2007)	95 668 hab.
0-14 ans	16 317 hab.
15-24 ans	11 977 hab.
25-44 ans	26 437 hab.
45-64 ans	29 048 hab.
65 ans et plus	11 889 hab.
Solde migratoire total (2006-2007)	- 953 hab.
Perspectives démographiques (variation 2026/2001)	- 18,1 %
Emplois (octobre 2008) ^{1,2}	48,6 k
Taux d'activité (octobre 2008) ^{1,2}	61,5 %
Taux d'emploi (octobre 2008) ^{1,2}	54,7 %
Taux de chômage (octobre 2008) ^{1,2}	11,2 %
Revenu disponible par hab. (2007)	25 438 \$
PIB aux prix de base (2007)	4 634,4 M\$
Dépenses en immobilisation (2008)	1 056 230 k\$
Exportations de marchandises (2005) ²	2 898,4 M\$

1. Désaisonnalisés en moyennes mobiles de trois mois.

2. Données se rapportant aux régions de la Côte-Nord et du Nord-du-Québec.

Imprimer